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SCIENTIFIC CRITERIA DOCUMENT FOR THE DEVELOPMENT OF PROVINCIAL WATER QUALITY GUIDELINES FOR ORGANOTINS

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FOR THE DEVELOPMENT OF
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FOR ORGANOTINS

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SCIENTIFIC CRITERIA DOCUMENT FOR THE DEVELOPMENT OF PROVINCIAL WATER QUALITY GUIDELINES FOR ORGANOTINS

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considerably lower than those of other agencies that do not use this approach. Guidelines can be promoted to Objectives when sufficient reliable data become available.

PWQO/Gs are used to designate surface waters of the Province which should not be further degraded. They are also used in receiving water discharge assessments and may be included in Certificates of Approval which are issued to regulate effluent discharges. Where better water quality is required to protect other beneficial uses of the environment in a given location, appropriate criteria and factors, including public health considerations, are taken into account.

SUMMARY

All organotin compounds currently used in Canada are now imported. Approximately one million kilograms were used annually in Canada in the early 1980s. No information was found on amounts used in Ontario. About 97% of organotins are used in a variety of industrial applications including stabilizers for polyvinyl chloride polymers, catalysts in production of polyurethane foam, additives in lubricants and oils, fire retardants, and dielectrics in transformers. About 2-3% are used as biocides, primarily as antifouling agents in marine paints. The chipping or leaching of marine paints on the hulls of boats represents the principal route of entry of organotin compounds into the aquatic environment. There are currently no organotins registered for use as agricultural pesticides in Ontario.

Tributyltin (TBT) and its breakdown products, dibutyltin (DBT) and monobutyltin (MBT) are moderately persistent in subsurface water and undisturbed sediments with a half-life in the order of 6 months to a year. Little information is available on the persistence of other organotin species. Levels of organotins in the natural environment are highly variable. In an Ontario-wide survey, concentrations of TBT in subsurface waters ranged from undetectable to 2.34 µg/L Sn found in Port Hope harbour. Detectable levels of butyltins were found in 31 of 73 sediment samples with a high concentration of 4.73 µg/g Sn for monobutyltin in Lake Nipissing.

The toxicity of the organotins to aquatic organisms varies among compounds, with the triorganotins showing the greatest toxic effects. Tributyltin and triphenyltin are the most toxic species of the organotins for which information is available. Tributyltin is moderately persistent in both water and sediments with a half-life in the order of six months to a year. Little information is available on the persistence of other organotin species.

There are many studies on the toxicity of triorganotins in the freshwater environment but the majority of these are acute. Of those reporting chronic effects, some studies found that serial dilutions of a measured concentration were still toxic at concentrations below analytical detection limits, (eg. the TBT data, shown below). In those cases, the lowest concentrations causing observed chronic toxic effects could only be reported as nominal

concentrations which resulted in the data being classified as secondary. This classification does not necessarily indicate poor experimental technique.

A review of the literature revealed that at present, sufficient data exist to set Provincial Water Quality Guidelines (PWQGs) for di-*n*-butyltin, tributyltin, triethyltin and triphenyltin. The lowest concentrations that caused observed effects for di-n-butyltin (DBT) occurred at 29.4 µg as DBT ion (decreased weight and metabolic affects in duck mussel); for triethyltin (TET) at 85 µg/L as TET ion (arrested larval development in sturgeon); for tributyltin (TBT) at 0.00097 µg/L as TBT ion (reduced oviposition in snails); and for triphenyltin (TPT) at 0.21 µg/L as TPT ion (25% mortality, and reduced resistance of survivors to infection, in rainbow trout yolk sac fry). In each case an uncertainty factor was applied to these values to account for the scarcity of data.

The PWQGs that have been determined for these organotin species are as follows:

Species	Critical Value (µg/L)	Uncertainty factor	PWQG (µg/L)
Di-*n*-butyltin	29.4 DBT^{2+}	357	8×10^{-2} as DBT ion
Triethyltin	85 TET^{+}	203	4×10^{-1} as TET ion
Tributyltin	0.00097 TBT^{+}	181	5×10^{-6} as TBT ion
Triphenyltin	0.21 TPT^{+}	103	2×10^{-3} as TPT ion

The PWQG for tributyltin is less than the current detection limit in water (1 µg/L as Sn in 1 litre samples for routine analyses; 1×10^{-3} µg/L as Sn in 1 litre samples, in analyses for research). Primary chronic toxicity data are required for all these organotins in order to reduce their uncertainty factors and permit development of Provincial Water Quality Objectives.

These guidelines can be promoted to objectives when sufficient primary data become available. The data required are outlined in a research needs section in the PWQG document. This will allow a "safety factor" to be used in the development process instead of the "uncertainty factor" which reflects the potential of the toxicant to bioaccumulate and the quantity of high quality data available.

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The public was notified of the proposed Provincial Water Quality Guidelines for Organotins through the Environmental Bill of Rights electronic registry and given the opportunity to comment in accordance with the Environmental Bill of Rights.

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Notes and Abbreviations Used in the Text

Abbreviations for organotin compounds used in this document are as follows:

MBT = monobutyltin
DBT/DBT+ = dibutyltin ion(s)
DBTC = di-*n*-butyltin dichloride
DBTT = di-*n*-butyltin trioxide
TBT/TBT+ = tributyltin ion(s)
TBTC = tributyltin chloride
TBTO = tributyltin oxide
TET/TET+ = triethyltin ion(s)
TETB = triethyltin bromide
TETC = triethyltin chloride
TPT/TPT+ = triphenyltin ion(s)
TPTC = triphenyltin chloride
TPTH = triphenyltin hydroxide

Concentrations in this document are expressed in a number of different units commonly used in scientific papers. The conversion factors are:

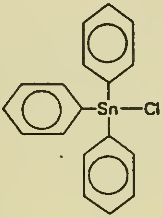
1 gram per litre (g/L) = 1000 milligrams per litre (mg/L)
1 mg/L = 1000 micrograms per litre (µg/L)
1 µg/L = 1000 nanograms per litre (ng/L)

The term LC50 denotes the concentration of a chemical in the water resulting in 50% mortality of a test population. In this report, LC100 signifies the lowest exposure concentration tested causing 100% mortality.

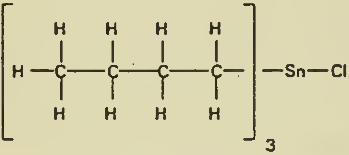
1 INTRODUCTION

1.0 Introduction and Background

Organotins are organometallic compounds - compounds in which the cation is formed by a carbon atom of one or more of a specific class of organic molecules bound to a metal atom and the associated anion can be any one of a variety of anions such as chlorides or hydroxides. The various classes of organotins include dibutyltins, triethyltins, tributyltins, and triphenyltins. Examples of structural formulae of organotins are shown below.



Triphenyltin chloride



Tributyltin chloride

The properties of these organotins vary with the number and length of the hydrocarbon chains. They are normally salts at room temperature, non volatile, and slightly soluble in water, but because of the organic moiety, the water solubility decreases further with increases in total chain length. Conversely, the lipid (fat) solubility increases to moderate levels with increases in total chain length. The log Kow's, toxicity, and potential to bioaccumulate of TBT and TPT are therefore greater than those of TET and DBT. However, the organic portion of each molecule gives them all the tendency to associate with sediment particles, and any hydrophobic molecules in the aquatic surface micro-layer. Tributyltins are moderately persistant in both water and sediments with a half-life in the order of 6 months to a year. Little information is available on the persistence of other organotin species.

1.1 Manufacture and Use of Organotins in Canada

Although dibutyltin dilaurate was formerly manufactured in Canada, all organotin compounds currently used in Canada are imported (Maguire 1991).

Organotin compounds have been used for a variety of industrial and agricultural applications in Canada (Jones and Millson 1982; Thompson et al. 1985). The largest proportion of organotins, about 80-82% (Jones and Millson 1982), is used to stabilize polyvinylchloride (PVC) polymers to maintain both clarity and impact strength (Luijten 1972). About 16-17% of organotins are used as catalysts in the production of polyurethane foams. Minor industrial uses include their use as additives in lubricants and oils, fire-proofing agents for plastics and woods, antioxidants and dielectrics in transformers. About 2-3% of organotins have been used as biocides, primarily as antifouling agents in marine paints, as well as fungicides, bactericides, insecticides and acaricides in agriculture. In Ontario, there are currently no organotins registered for use as agricultural pesticides in Ontario. The Canadian government regulated the use of tributyltin under the Pest Control Products Act in 1989. The maximum daily release rate now permitted in Canada for TBT for use as an antifouling paint on boats is 4 $\mu g/cm^2$ of hull surface. Tributyltin is prohibited entirely on vessels less than 25 m in length.

A listing of major organotin compounds that have been used in Canada and their applications are presented in Table 1.1. Jones and Millson (1982) reported that Canadian consumption of organotins was dominated by about 10 organotin compounds . They also reported that the total annual Canadian market for organotins exceeded 10^6 kg, about 10% of that of the U.S. (~ 11 x 10^6 kg; Zuckerman et al. 1978).

1.2 Sources of Organotins in the Aquatic Environment

Little information is available on the routes of entry of organotins to the Canadian aquatic environment. Production losses of organotins from the one Canadian plant that formerly produced dibutyltin dilaurate are not available, although U.S. figures estimate that 0.5% of production is normally lost at the manufacturing source (Zuckerman et al. 1978). An estimate of the losses of organotins used in industrial processes is also unavailable.

Organotins used in crop protection have been employed in too small quantities to have been a significant source (Thompson et al. 1985). The greatest potential for release of organotins to the aquatic environment results from their use as biocidal agents (Lapp 1976). Organotins in marine paints and preservatives leach into the water from surfaces treated with these compounds. The occurrence of high concentrations of tributyltin (TBT) in water and sediments from areas with high boating activity has been attributed to use of TBT in antifouling paints applied to boats (Maguire et al. 1982; Maguire 1984; Clark et al. 1988; Maguire et al. 1986).

Table 1.1 Organotin Compounds Used in Canada (from Jones and Millson 1982).

Compound	Uses
Dibutyltin bis-(isooctylmercaptoacetate)	Stabilizer for PVC plastics used in siding, eavestroughs and soffits
Dibutyltin dilaurylmercaptide	Polyurethane foam catalyst; feed additive
Dibutyltin oxide	Precursor for a dibutyltin dilaurate
Dibutyltin dilaurate[a]	Chicken feed additive; catalyst for urethanes; esterification catalyst
Bis(tributyltin) oxide	Slimicide in cooling water towers; antifouling additive for marine paint; preservative
Tributyltin fluoride	Antifouling additive for marine paint
Dioctyltin maleate	Stabilizer for rigid PVC pipe for potable water supplies
Dibutyltin diacetate	Catalyst for flexible foams
Dioctyltin bis-isooctyl mercaptoacetate	Stabilizer for plastics used in food packaging
Di[tri-2-methyl-(2-phenylpropyl)tin] oxide	Acaricide (crop protection)[b]
Tricyclohexyltin hydroxide	Fruit protection [b]
Triphenyltin hydroxide	Fungicide for potatoes[b] (use discontinued in 1980)

[a] Formerly manufactured in Canada

[b] No organotins are currently registered for use as pesticides in Ontario.

Biological methylation has also been suggested as a possible source of organotins to the aquatic environment. Methylation of tin or butyltins has been demonstrated in nutrient media and mixtures of natural water and sediment (Cooney 1988; Guard et al. 1981; Chau et al. 1981). Methyltins have been found in a variety of natural environments including Ontario lakes and rivers (Maguire 1984; Maguire et al. 1986). The presence of mixed methylbutyl species of alkyltins in water and sediments has been suggested as additional proof of natural methylation since it is unlikely that these compounds are of anthropogenic origin (Maguire 1984; Maguire et al. 1985; Maguire et al. 1986; Maguire 1987). Methyltin species, however, have not been detected consistently in water and sediments and are usually present in much smaller concentrations than TBT or its degradation products. Methylation of tributyltin does not appear to be a significant pathway of transformation (Maguire 1987).

1.3 Concentrations of Organotins in Ontario Waters

The occurrence of organotin compounds, particularly the butyltins and methyltins in Ontario waters has been studied extensively (Maguire et al. 1982; Maguire et al. 1985; Maguire and Tkacz 1985, 1987; Maguire et al. 1986; Maguire 1987). Table 1.2 from Maguire et al. (1986) presents measurements of butylated tin species in Ontario waters and sediments sampled during a Canada-wide survey (1982-1985). At Port Hope, for example, concentrations of TBT in sub-surface waters ranged from non-detectable to 2.34 µg/L Sn (Table 1.2). Detectable levels of TBT were found in 33 of 98 samples of subsurface water. Consistent with its use in antifouling paints, TBT was found in areas of heavy boating or shipping traffic. The concentration obtained at Port Hope is possibly the highest level of TBT ever measured (Clark et al. 1988). In 18 of the 33 locations at which detectable levels of TBT were determined, concentrations of TBT exceeded levels shown to be chronically toxic to early life stages of rainbow trout (0.07 µg/L Sn ; Seinen et al. 1981). Concentrations of dibutyl and monobutyltins in Ontario subsurface waters ranged from non detectable to 1.36 µg/L Sn at Belleville and 1.89 µg/L Sn at Port Stanley, respectively.

Detectable levels of butyltins were found in 31 of 73 sediment samples (Table 1.2). In general, the pattern of occurrence of butyltins in sediments was similar to that in water. The highest concentration of a butyltin species in Ontario sediments was 4.73 µg/g Sn for

Table 1.2 Concentration of butyltin species in subsurface water and surficial sediments at Ontario locations. Blanks indicate levels less than detection limits* (modified from Maguire et al. 1986).

Location	Water Concentration (µg/L Sn)			Sediment Concentration (µg/g Sn)		
	TBT	DBT	MBT	TBT	DBT	MBT
Wabigoon R.	0.01	0.12	0.25			
Clay L.(1)			0.19			0.07
Clay L.(2)			0.03			
Wabigoon R.			0.15	0.04		0.03
Thunder Bay(1)	0.01	0.02		0.01		
Thunder Bay(2)	0.08	0.01				
Thunder Bay(3)		0.07	0.15			
Nipigon R.	0.64					
Terrace Bay	0.49					
Marathon (1)			0.05			
Marathon (2)	0.02					
Turkey L.(1)	0.09					
Turkey L.(2)	0.05	0.01	0.06			
Turkey L.(3)					0.07	0.09
Turkey L.(4)	0.08					
St. Mary's R.(1)	1.68	0.09				
St. Mary's R.(2)	0.11					
Blind R.	0.57					0.02
Elliot L.			0.01			
Simon L.				0.01	0.01	
Kelley L.				0.03		
Ramsey L.	0.02					0.22
Nepewassi L.	0.04	0.01	0.01	0.02		0.24
Ashigami L.		0.93	0.03			0.32
L. Nipissing at North Bay					0.58	4.73
L. Muskoka	0.04				0.21	2.5
L. Simcoe at Barrie						0.04
St. Clair R. (1)		0.05				
St. Clair R. (2)	0.22		0.03	0.01		
St. Clair R. (3)	0.22					
St. Clair R. (4)						
L. St. Clair (1)	0.1		0.01			
L. St. Clair (2)		0.04	0.03			
L. St. Clair (3)	0.07			0.04	0.01	
Detroit R.			0.01			
Port Stanley	0.29	0.2	1.89	0.01		
Port Dover	0.07	0.01		0.01	0.01	
St. Catherines (1)	0.03					
St. Catherines (2)		0.01				
Port Weller				0.01	0.01	
Credit R.	0.03			0.01		
Toronto Harbour (1)	0.03	0.01	0.59	0.04		0.07
Don R.				0.02	0.03	0.01
Toronto Harbour (2)	0.24	0.01	0.11		0.01	0.09
Whitby (1)	1.72	0.74	0.42	0.16	0.22	0.4
Whitby (2)	0.1	0.06	0.01			
Port Hope	2.34	0.04	0.28			
Cobourg		0.01		0.02		0.02
Moira L.				0.03	0.15	0.27
Moira R.	0.01					
Belleville	0.01	1.36				
Kingston Harbour					0.15	0.21
St. Lawrence R. at Maitland	0.05		0.01			
St. Lawrence R. at Cornwall (1)	0.01				0.02	0.03
St. Lawrence R. at Cornwall (2)				0.01		
Ottawa R. at Chalk River	0.52	0.02				
Ottawa R. at Ottawa				0.09	0.03	

* Detection limits were 0.003 µg/L Sn and 0.003 µg/g Sn for water and sediment respectively.

monobutyl tin found in Lake Nipissing. The toxicological significance of these levels in the sediments is unknown. Also unknown is the extent to which paint chips in the sediment contribute to measurements of TBT in sediment samples (Maguire 1987).

1.4 Concentrations of Organotins in Freshwater Biota

There are as yet few published studies of organotin residues in European or North American freshwater organisms. In the survey of Ontario lakes by Maguire et al. (1986), analysis of 17 individual Ontario sport fish yielded only six fish with significant levels of organotins (primarily methyltin species). The highest concentration was 0.88 µg/g Sn of monomethyl tin found in lake trout from Lake Ontario at Port Credit. Consistent with findings in waters and sediments, tributyltin was found only in fish from harbours with heavy boating activity.

1.5 Fate of Organotins in the Aquatic Environment

Little reliable information is available on the environmental fate of the majority of organotins in the aquatic environment. Since butyltins have been studied most intensively, discussions will concentrate on these species, especially TBT. Tri-*n*-butyl tin dissolves readily in contact with water. The aqueous solubility of Bis-(tri-*n*-butyltin) oxide (TBTO), ranging in the literature from 0.7 - 7 mg /L (Maguire et al. 1983), is far above acute toxicity levels. On dissolution, tributyltin oxides, salts and hydroxides form equilibrium mixtures of the initial compound, the chloride salt, the carbonate and the solvated ion, the relative amounts being a function of pH (Guard et al. 1982; Laughlin et al. 1986). Unlike the tetraalkyl tins or tetraalkyl leads, TBT is non-volatile with a low vapour pressure (6.4×10^{-3} mm Hg for TBTO). In laboratory tests, losses of TBT through volatilization were immeasurable over a period of 62 days (Maguire et al. 1983).

Once in solution organotins associate strongly with hydrophobic molecules in the surface microlayer (Maguire et al. 1982, 1985; Hall et al. 1987; Maguire and Tkacz 1987; Batley et al. 1989). Concentration factors of up to 10^4 have been measured for a variety of alkyltin species in the surface microlayer relative to the subsurface water (Maguire et al. 1982; Maguire and Tkacz 1987). The highest concentration of alkyltins in the surface microlayer (473 µg/L Sn) was found in the Moira River near Belleville (Maguire and Tkacz 1987). Occasionally, concentrations in the surface microlayer were significant relative to total amounts in the subsurface water column.

For example, at Belleville, the amount of TBT in the microlayer was equivalent to 71% of the total amount in the whole depth of subsurface water (actual depth unreported). Association of organotins with surface microlayers may have implications for organisms living at the air-water interface, for accurate determination of the amounts of the organotins present and for the possibility of enhanced photolysis (see below). However, because the surface microlayer is affected by turbulence and the presence of surface active material, concentrations in the microlayer are extremely variable with time and serve as poor indicators of the subsurface water concentrations to which most aquatic organisms are exposed (Maguire et al. 1985).

Organotins also associate strongly with particulate matter. In laboratory experiments, Maguire and Tkacz (1985) found that TBT bound strongly to sterile Toronto Harbour sediments and, if left undisturbed, no desorption was evident over a ten month period. However, if the sediments were agitated, the rate of desorption was rapid. For example, shaking a sediment-water mixture for three days released 80% of sediment-bound TBT. This observation is consistent with those of Unger et al. (1988) and Unger (1989) who showed that sorption of TBT on freshwater and estuarine sediments in the laboratory was both rapid and reversible. While sorption isotherms of TBT on sediments were linear, desorption kinetics indicated an initial fast rate of desorption followed by a slower rate. The sediment-water partition coefficient was determined as 3×10^3 (Maguire et al. 1986; Cardwell and Sheldon 1986). The biological availability of sediment residues of TBT has received little attention. One study has shown that oligochaetes are capable of assimilating TBT from sediments and degrading it to inorganic tin (Maguire and Tkacz 1985).

Organotins are moderately persistent in water and sediments. They are subject to degradation through chemical, photolytic and biological mechanisms. Chemical degradation is a fairly slow process with a measured half-life for TBT of greater than 11 months in a fresh water sediment mixture (Maguire and Tkacz 1985). Photolysis in sunlight is a more rapid mechanism of degradation although still fairly slow. In laboratory experiments, TBT has been shown to decompose by successive debutylation in the presence of ultraviolet light (Blunden and Chapman 1986; Thompson et al. 1985). Maguire et al. (1983) found that degradation of TBT in Hamilton Harbour water exposed to sunlight at 20°C occurred with a half-life of 89 days. Dibutyltin, monobutyltin and inorganic tin accounted for 75% of the loss of TBT. Little loss of TBT was observed in the dark. Dibutyltin and monobutyltin decomposed more rapidly than TBT with half-lives of 9 days and 0.4 days respectively. Photolysis was enhanced in the presence of dissolved organic material such as fulvic acid. Because of the slowness of the photolytic process and the normal attenuation of light through the water column, photolysis of TBT is probably unimportant at depths greater than 0.5-1 m (Maguire 1987), although the effects of sunlight on TBT associated with the surface microlayer may be considerable. In contrast to TBT, triphenyltin was observed to degrade rapidly in sunlight with a loss of 70% in 36 days (Soderquist and Crosby 1980). Breakdown products were diphenyltin, monophenyltin and inorganic tin. No breakdown of triphenyltin was observed in the dark.

Biological degradation in water and sediment appears to be the most important mechanism limiting the persistence of TBT in aquatic ecosystems (Maguire 1987). TBT in Toronto Harbour water and a Toronto Harbour sediment-water mixture was degraded by bacteria in the dark at 20°C with half-lives of 140 ± 35 days and 112 ± 14 days, respectively (Maguire and Tkacz 1985). TBT is therefore expected to persist in the Ontario waters with a half-life of about 5 months (water) and 4 months (sediments).

Because of the relative importance of biological degradation, ecological factors such as temperature or the presence/absence of the appropriate microorganisms will play a significant role in determining the persistence of tributyltin in any particular location. For example, degradation of TBT by microorganisms from Toronto Harbour was about twice as fast under anaerobic conditions as under aerobic conditions (Maguire et al. 1985). At the cooler temperatures more characteristic of Ontario waters, the half-lives quoted above (5 and 4 months) may be effectively doubled (Maguire and Tkacz 1985).

2 TOXICITY OF ORGANOTINS TO AQUATIC ORGANISMS

Large differences have been observed in the toxicity of the various organotins to aquatic organisms. In general, the triorganotin compounds have been shown to be much more toxic than any other group. Biological activity increases with side-chain size up to a length of 5-6 carbons, then declines again (Laughlin and Linden 1985; Thompson et al. 1985). In studies on the toxicity of various di and tri organotins to the late larval stage of mud crabs, Laughlin et al. (1984) found that the hydrophobicity and the partitioning behaviour of an organotin control its uptake and thus the lethal exposure level for aquatic organisms.

The toxicity of organotins (primarily butyltins) to aquatic organisms has been extensively reviewed (see Hall and Pinkney 1985; Thompson et al. 1985; Eisler 1989). However, since these studies largely involve marine organisms, the majority are not directly relevant to the setting of Ontario provincial guidelines or objectives.

Sufficient data exist only for the setting of guidelines for di-*n*-butyltin, tributyltin, triethyltin and triphenyltin. Because of the scarcity of relevant toxicity studies, insufficient data are available to draw conclusions on the effects of variables such as pH, alkalinity and hardness on the toxicity of these compounds.

In presenting the toxicity studies it was assumed that the dissolution of each organotin in water produces the hydrated cation, regardless of the initial species. For example, all tributyltin compounds Bu_3SnX, where X = F, Cl, Br, acetate or $OSnBu_3$ (for the bis compound), yield the same toxic species, Bu_3Sn^+. This assumption permits comparison of a large number of toxicity studies in which different chemical species have been used. Compound concentrations have been expressed with the concentration of the ion in parentheses. This usage of a common organotin cation, also proposed by Maguire (1987) and Maguire et al. 1985), is supported by observations that chromatographic behaviour, metabolism and biological activity are independent of X. Additional support comes from chemical studies indicating that the organotin hydroxides or bis-oxides form equilibrium mixtures of the starting material and common anions (see Section 1.4). Tabulations of physical/chemical properties for each organotin (Appendices I-IV, Table 1) have been made on the most common compound(s).

In order to derive water quality objectives and guidelines, the Ontario Ministry of the Environment and Energy distinguishes between primary data and secondary data (MOE 1992). In studies yielding primary data, test concentrations and environmental parameters (pH, hardness etc.) are monitored through each experiment and standardized statistical and test procedures are followed. All studies reviewed in this report were judged as yielding primary or secondary data and the reasons for each decision are presented.

2.1 Acute Toxicity

2.1.1 Fish

2.1.1.1 Di-*n*-butyltin (DBT)

There are no primary data available on the acute toxicity of di-*n*-butyltin (DBT) to freshwater fish. The only available study is that of Blunden and Chapman (1986) who report a 48h-LC_{50} for golden orfe (Leuciscus idus melanotus) of 1000 µg/L di-*n*-butyltin dichloride (DBTC) (766 µg/L DBT^+). Since test concentrations were not measured, the data are considered secondary.

2.1.1.2 Triethyltin (TET)

As with DBT, very few studies have investigated the acute toxicity of triethyltin to freshwater fish and all yield secondary data. Stroganov et al. (1974) exposed carp to triethyltin chloride (TETC) at 1000 µg/L (853 µg/L TET^+) for 2 and 6 days. Decreased activity, impaired respiratory rhythm and loss of equilibrium were observed within four days. All carp died within 7 days. Buzinova (1977) observed 100% mortality of yearling carp exposed for 3-5 days to TETC at 500 µg/L (426 µg/L TET^+). Test concentrations were nominal in both studies and the data are secondary.

2.1.1.3 Tributyltin (TBT)

A significant number of studies have been conducted on the acute toxicity of TBT to freshwater fish. These studies involve nine freshwater species of which rainbow trout is the most common test organism. All but two of these studies (those by Brooke et al 1986 and Martin et al 1989) were considered secondary because nominal test concentrations were used.

The earliest study on rainbow trout is that by Alabaster (1969) who reported 24 and 48h-LC_{50}'s of 28 and 21 µg/L TBTO (27.2 and 20.4 µg/L TBT^+). Chilamovitch and Khun (1977) observed behavioral changes in rainbow trout exposed to TBT including lethargy and loss of positive rheotaxis. They calculated a 24h-EC_{50} of 30.8 µg/L TBTO (30.0 µg/L TBT^+) based on the loss of positive rheotaxis. Between 80-90% of the fish died post-treatment. Additional acute affects in this study included damage to the gill epithelium and cytological abnormalities during a 21.5h exposure to 58 µg/L TBTO (56.4 µg/L TBT^+) and damage to the bile duct and cornea after 5 to 7d exposure to 11.7 µg/L TBTO (11.4 µg/L TBT^+). In other studies on rainbow trout, Blunden and Chapman (1986) and Douglas et al. (1986) obtained 96h-LC_{50}'s ranging from 11.2 to 20 µg/L TBTO (10.9 to 19.4 µg/L TBT^+). Brooke et al. (1986) observed loss-of-equilibrium and mortality in juvenile rainbow trout with a 96h-EC_{50} and 96h-LC_{50} of 3.9 µg/L TBT^+ for each endpoint, respectively. In a recent study, Martin et al. (1989) reported a 96h-LC_{50} for rainbow trout of 1.41 µg/L Sn (3.44 µg/L TBT^+).

In studies on fathead minnows, Argaman et al. (1984) reported a relatively high 96h-LC_{50} for TBTO of 50-200 µg/L (49 to 195 µg/L TBT^+). Brooke et al. (1986), however, found this species to be more sensitive to TBTO with a 96h-LC_{50} of 2.6 µg/L TBT^+.

For guppies, an early study using TBTO by Floch et al. (1964) obtained a 24h-LC_{100} of 75 µg/L (73 µg/L TBT^+). Schantzberg and Harris (1978) reported a range of 24 h-LC_{50}'s from 10 to 20 µg/L TBTO (9.7 to 19.5 µg/L TBT^+).

In acute studies using other fish species, the 96h-LC_{50} obtained for the channel catfish was 5.5 µg/L TBT^+ (Brooke et al. 1986), for the lake trout was 12.7 µg/L TBT^+ (Martin et al. (1989), and for the bluegill (Lepomis macrochirus) was 240 µg/L TBTO (234 µg/L TBT^+; Foster 1981). Matthiessen (1974) obtained a 24h-LC_{50} for the Mozambique mouthbrooder (Tilapia mossambica) of 28 µg/L TBTO (27 µg/L TBT^+) with a lethal threshold between 8 and 16 µg/L TBTO (7.8 and 15.6 µg/L TBT^+). Chilamovitch and Khun (1977) observed the same behavioral effects on Tilapia rendalli as on rainbow trout exposed to TBTO (see above) and calculated a 24h EC_{50} of 53.2 µg/L TBTO (51.7 µg/L TBT^+). Other miscellaneous studies include those by Deschiens et al.(1966) on the jewelfish, Tilapia nilotica (24h-LC_{70} = 45 µg/L TBTO; 43.7 µg/L TBT^+) and Plum (1981) on the golden orfe, Leuciscus idus melanotus, (48h-LC_{50} = 50 µg/L TBTO; 48.7 µg/L TBT^+)

2.1.1.4 Triphenyltin (TPT)

As with tributyltin, a considerable number of studies have reported on the acute toxicity of triphenyltin (TPT) to freshwater fish. Many of the early studies were conducted in the 1960's during a period of intensive research to determine the potential of triorganotins as molluscicides in the control of schistosomiasis. Typically, these studies utilized nominal test concentrations and static exposures without renewal and were therefore considered secondary. During this period, the acute toxicity of triphenyltin acetate was determined by Floch and Deschiens (1962), Floch et al. (1964), Gras and Rioux (1965), Alabaster (1969), and Knauf (1974) on at least eight freshwater fish species including carp (Cyprinus carpio), goldfish, guppies (Poecilia reticulata), mosquitofish (Gambusia affinis), harlequin fish (Rasbora heteromorpha), eel, golden orfe, Leuciscus idus melanotus, and rainbow trout. Toxicity values expressed as 24-48h LC_{50}'s and LC_{100}'s ranged from 46 to 1030 µg/L TPT^+ (see Appendix IV, Table 2 for details). These values are generally higher than those obtained in later studies.

In more recent studies using triphenyltin hydroxide (TPTH), Tooby et al. (1975) reported a 96h-LC_{50} of 15 µg/L TPTH (14.3 µg/L TPT^+) for rainbow trout and a 48h-LC_{50} of 42 µg/L TPTH (40 µg/L TPT^+) for harlequin fish. Mayer and Ellersieck (1986) reported 24h and 96h LC_{50}'s for rainbow trout (95 and 26.7 µg/L TPT^+), goldfish (115 and 59.1 µg/L TPT^+), fathead minnow (72 and 19.1 µg/L TPT^+), and bluegill (72 and 21.9 μg/L TPT^+). Jarvinen et al. (1988), exposing fathead minnow larvae to TPTH for 12, 24, 48, and 72h, followed by transfer to control water for the remainder of a 96 h test, observed latent mortalities with LC_{50}'s ranging from 61.8 to 6.0 μg/L TPTH (59 to 5.7 μg/L TPT^+) and EC_{50}'s ranging from 50.3 to 3.5 μg/L TPTH (48 to 3.3 μg/L TPT^+) where surface swimming was the behavioral endpoint. Exposure to TPTH for the full 96h exposure period resulted in a mean LC50 of 7.1 μg/L (6.8 μg/L TPT^+) and a mean EC50 of 3.7 μg/L (3.5 μg/L TPT^+). Of these recent studies, only that by Jarvinen et al. (1988) was considered primary. The rest of the studies used nominal test concentrations or provided insufficient information on which to judge the quality of the data.

2.1.2 Invertebrates

2.1.2.1 Di-*n*-butyltin

Vighi and Calamari (1985) present the only available study on the acute toxicity of dibutyltin dichloride (DBTC) to freshwater invertebrates. In a 24h acute immobilization test (OECD 1981), the IC_{50} for Daphnia magna was determined as 900 μg/L (690 μg/L DBT^+). This study, using nominal test concentrations, was considered secondary.

2.1.2.2 Triethyltin

Vighi and Calamari (1985) also exposed young (< 24h old) Daphnia magna to triethyltin bromide (TETB) in the 24h acute immobilization test. The resulting 24h-IC_{50} was 260 μg/L TETB (187 μg/L TET^+). No other studies on the toxicity of triethyltin to invertebrates are available. As above, the study by Vighi and Calimari (1985) was considered secondary.

2.1.2.3 Tributyltin

Early studies on the acute toxicity of TBTO and tributyltin acetate to invertebrates concentrated on species of the snail Biomphalaria in research on schistosomiasis. Results of these studies, those on other related molluscs, on the cladoceran Daphnia longispina, and on the ostracod Cypridopsis have been summarized in Hall and Pinkney (1985). Six to 48 hour exposures of Biomphalaria to TBT with recovery periods lasting up to 72 h, yielded LC_{50}'s ranging from 8.3 to 399 μg/L TBT^+ and LC_{100}'s ranging from 42 to 1900 μg/L TBT^+ (Ritchie et al. 1964; Camey and Paulini 1964; Frick and Jimenez 1964; Hopf and Muller 1962; Seiffer and Schoof 1967).

Also in Biomphalaria, 1-5d exposures with recovery periods lasting up to 72 h caused 100% mortality at 25 to 250 μg/L TBT$^+$ (Floch et al. 1964). In Daphnia longispina, 24-72h LC$_{100}$'s ranged from 58-249 μg/L TBT$^+$(Floch et al. 1964; Deschiens and Floch 1968) while 24-96h LC$_{100}$'s for the ostracod Cypridopsis hartwigi ranged from 116 to 3900 μg/L TBT$^+$ (Floch et al. 1964; Deschiens and Floch 1968; see Appendix III Table 2 for details). Foster (1981) reported a 24h LC$_{50}$ of 2100 μg/L TBTO (2044 μg/L TBT$^+$) for the Asiatic clam larva Corbicula fluminea. In a study by Kumar Das et al. (1984), 4th and 2nd instar larvae of the mosquito Aedes aegypti were exposed to a variety of organotin compounds including tributyltin chloride and TBTO. The 24h-LC$_{50}$ for tributyltin chloride on 4th instar larvae was 390 μg/L (347 μg/L TBT$^+$) while the 24h-LC$_{50}$'s for TBTO on 4th and 2nd instar larvae were 380 μg/L (370 μg/L TBT$^+$) and 17 μg/L (16.5 μg/L TBT$^+$), respectively. In subsequent delayed-effect studies, 2nd and 4th instars were exposed to their approximate LC$_{50}$ concentrations of TBTO and TBTC for 24 hours then transferred to clean water for 5 to 8 days. Mortality of the 4th instar larvae was 55.0% during the 24h exposure to 350 μg/L TBTC (312 μg/L TBT$^+$) and 55.5% during the 24h exposure to 300 μg/L TBTO (292 μg/L TBT$^+$). Post-treatment mortality was 25% and 27% respectively. Post-treatment mortality of second instar larvae was 16% after exposure to 17 μg/L (16.5 μg/L TBT$^+$). In all of these earlier studies nominal exposure concentrations were used and the data are considered secondary.

More recent studies have yielded lower toxicity levels of TBT to invertebrates. Median immobilization concentrations (24h IC$_{50}$s) of 13 μg/L TBTC (11.6 μg/L TBT$^+$) (Vighi and Calamari (1985)) and 68 μg/L TBT$^+$ (Foster 1981) were reported for Daphnia magna. Meador (1986) obtained a 96h-LC$_{50}$ of 5.9 μg/L TBTC (5.3 μg/L TBT$^+$) for this species. Brooke et al. (1986) reported acute toxicity levels of TBTO to a variety of invertebrates. Forty-eight or 96h IC$_{50}$'s were measured on a coelenterate (Hydra sp., 0.5 μg/L TBT$^+$); an annelid (Lumbricus variegatus, 5.4 μg/L TBT$^+$); an amphipod (Gammarus pseudolimnaeus, 3.7 μg/L TBT$^+$); a mosquito (Culex sp.,10.2 μg/L TBT$^+$) and a cladoceran (Daphnia magna, 4.3 μg/L TBT$^+$). Of these recent studies only that by Brooke et al. (1986) is considered primary, the rest reporting only nominal test concentrations.

2.1.2.4 Triphenyltin

As with TBT, earlier studies on the acute toxicity of triphenyltin to invertebrates were concentrated on species of snails including Biomphalaria and Bulinus and generally yielded toxicity values one to three orders of magnitude greater than those found in later studies (Hall and Pinkney 1985). For example, 24h-LC$_{50}$'s on Biomphalaria ranged from 43 μg/L TPT^+ (Hopf et al. 1967) to 908 μg/L TPT^+ (Deschiens and Floch 1968). The 24h-LC_{50}'s for young and adult Bulinus were 10.2 and 30 μg/L TPT^+, respectively (Webbe and Sturrock 1964).

In more recent studies, Mayer and Ellersieck (1986) reported a 96h-LC_{50} of 66 μg/L TPTH (63 μg/L TPT^+) for the amphipod Gammarus fasciatus. Vighi and Calamari (1985) obtained a 24h-IC_{50} of 19 μg/L TPTH (17.2 μg/L TPT^+) for <24h old Daphnia magna. Kline et al. (1989) reported 48h-IC_{50}'s of 14.5, 16.5 and 11.3 μg/L TPTH (13.8, 15.7 and 10.8 μg/L TPT^+) for Daphnia magna, D. pulex, and Ceriodaphnia dubia, respectively. In the study by Kumar Das et al. (1984) described above, 4th and 2nd instar larvae of the mosquito Aedes aegypti exposed to triphenyltin chloride exhibited 24h-LC_{50}'s of 420 μg/L (381 μg/L TPT^+) and 17 μg/L (15.4 μg/L TPT^+), respectively. In delayed-effect studies, 4th and 2nd instar larvae of the mosquito were exposed to their approximate LC_{50} concentrations for 24 hours then transferred to clean water for 5 to 8 days. Mortality of the 4th instar larvae was 54% during the 24h exposure to 400 μg/L triphenyltin chloride (363 μg/L TPT^+) and 46% post-treatment. Post-treatment mortality of the 2nd instar larvae was 45% after exposure to 17 μg/L (15.4 μg/L TPT^+). Of these studies only that by Kline et al. (1989) is considered primary. The remaining studies were considered secondary because they either reported nominal concentrations in static tests or provided insufficient information on which to determine the experimental conditions.

2.2 Chronic Effects

2.2.1 Fish/Vertebrates

2.2.1.1 Di-*n*-butyltin

De Vries et al (1991) exposed rainbow trout yolk sac fry to DBTC in a continuous flow system for 110 days from the time of hatching. A number of end points were observed including mortality, behaviour, body weight, organ weight, histopathologies, and resistance to bacterial challenge. At 800 nM DBTC (186 μg/L DBT^+) a mortality of 28% was observed over the entire exposure period. After ≈2 weeks, the fish became lethargic and usually swam in the lower half of the aquarium. Melanization of the tails was observed after 6-8 weeks. At the end of the exposure, body weights were reduced relative to controls and the ratios of liver mass to body weight were increased. Resistance of the survivors to infection by the bacterium (Aeromonas hydrophila) was decreased significantly. The NOEC for all endpoints was 160 nM (37 μg/L DBT^+). Since this study used only nominal concentrations, the data are considered secondary.

2.2.1.2 Triethyltin

Stroganov et al. (1974) examined the chronic effects of TETC on survival and carbohydrate metabolism of early juvenile (35-40 g) carp. Uptake of inorganic ^{14}C from the water, used as a metabolic indicator, was decreased in the brain, kidney, muscle and blood after a 40 day exposure of the fish to TETC at 10 and 100 μg/L (8.5 and 85 μg/L TET^+). Activity of the fish was slightly reduced at 85 μg/L TET^+ as were metabolic processes in the liver, brain and muscle. Glycogen formation was inhibited in the brain by 25% at 8.5 μg/L TET^+, and in the brain and liver by 45% and 20% respectively, at 85 μg/L TET^+.

Buzinova (1977) found that a 60 day exposure of yearling carp to 100 μg/L TETC (85 μg TET^+/L) significantly depressed (80%) digestive enzyme activity (amylase, trypsin, lipase). Exposure of carp to 3 μg/L TETC (2.6 μg/L TET^+) for 60 days resulted in an increase in intestinal amylase and lipase activity up to day 30, then a decrease to less than 50% that of the controls between days 30 and 60.

Danil'chenko (1977, 1982) examined the toxicity of TETC to the eggs, embryos and larval stages of several freshwater fish including the ruffe (Acerina cernua), loach (Misgurnus fossilis), sturgeon (Acipenser spp.), and pike (Esox lucius). In both studies exposure was continuous beginning at egg fertilization. The highest no-effect and lowest-effect concentrations for each developmental stage were noted. The sturgeon was the most sensitive species. Development of late yolk sac larvae was arrested at 85 μg/L TET^+ in the sturgeon, and 426 μg/L in the loach. In the ruffe and pike 100% mortality occurred at 850 μg/L TET^+. Because test concentrations in these studies were unmeasured and exact exposure times were not indicated in the two studies by Danil'chenko, these chronic studies are all considered secondary.

2.2.1.3 Tributyltin

Despite an abundance of literature on marine organisms, few relevant studies are available on the chronic toxicity of TBT to freshwater fish. In an early study, Polster and Halacka (1971) exposed the guppy Poecilia reticulata to a variety of TBT compounds for 7 days and observed LC_{50}'s for TBTO and TBTC of 39 and 21 μg/L (38 and 19 μg/L TBT^+). In one month and three month exposures of guppies (Poecilia reticulata) to TBTO, Wester and Canton (1987) observed a variety of histopathologies including thymus atrophy, hyperplasia of hemopoietic interstitial tissue of the kidney and liver vacuolation. Although evidence of these effects was noted at concentrations as low as 0.032 µg/L (0.031 µg/L TBT^+), severe thymus atrophy and liver vacuolation (the most sensitive endpoints) were noted at 1.0 µg/L (0.97 µg/L TBT^+) and 3.2 µg/L (3.1 µg/L TBT^+), respectively after the one month exposure. Reduced growth was also recorded at these concentrations (LOEC = 1.0 µg/L, NOEC = 0.32 µg/L). Both histological endpoints were observed at 0.32 µg/L (0.31 µg/L TBT^+) after the three month exposure but growth was not affected at this concentration. The histological NOEC for the three month exposure was 0.01 µg/L TBTO (0.0097 µg/L TBT^+). In separate experiments exposing freshly fertilized eggs of medaka fish (Oryzias latipes) to 3.1 µg/L TBT^+ for three months, Wester et al. (1990) observed histopathological effects in the liver, gas gland, kidney and retina. However, in this species, no thymus atrophy was found.

Brooke et al. (1986) exposed fathead minnow embryos and larvae to TBTO at concentrations ranging from 0.02 to 2.20 µg/L TBT^+ for 33 days with observable endpoints of hatchabilty, abnormal development, survival and growth. The lowest-observed-effect-concentration was 0.08 µg/L TBT^+ for reduced growth. The acute-chronic ratio for TBT ion calculated by the authors was 32.5.

Seinen et al. (1981) reported that 10-12d exposures of rainbow trout sac fry to 5 µg/L TBTC (4.5 µg/L TBT^+) caused 100% mortality suddenly during transition to the swim-up stage. During a 110d exposure, retarded growth (measured as weight), a reduction in haemoglobin concentration and erythrocyte count and diminished glycogen storage all occurred at concentrations as low as 0.2 µg/L TBTC (0.18 µg/L TBT^+). In the same study by de Vries et al (1991) noted above (Section 2.2.1.1), rainbow trout yolk sac larvae exposed to 0.6 nM TBTC (0.17 µg/L TBT^+) for 110 days showed a mortality of 25%, an increase in liver weight and decreases in body weight and liver glycogen content. A decrease in the resistance of the survivors to bacterial challenge was also observed. The NOEC for all five endpoints was 0.12 nM (0.035 µg/L TBT^+). Even at the NOEC the fish appeared lethargic and kept to the bottom half of the aquarium.

Laughlin and Linden (1982) studied the toxicity of TBT to embryos and larvae of the common European frog, Rana temporaria . Post gastrula embryos exposed for 5 days to 30 µg/L TBTO and TBTF (29 µg/L TBT^+) showed a significant decrease in percent body water (ratio of wet to dry weights). This effect level is considerably higher than most of the other chronic assays described above. Seinen et al. (1981) suggests that the gelatinous nature of the egg mass may have bound much of the available TBT ion during exposure.

Of these chronic studies on TBT, only those by Brooke et al. (1986) and Wester and Canton (1987) are considered primary. The other studies used unmeasured test concentrations.

2.2.1.4 Triphenyltin

In one of two available studies on the chronic toxicity of TPT^+ to freshwater fish, Jarvinen et al. 1988 reported significant decreases in growth and survival in newly-hatched fathead minnow larvae after a 30 day continuous exposure to concentrations of 0.23 μg/L TPTH (0.22 μg/L TPT^+) for growth, and 2.0 μg/L TPTH (1.9 μg/L TPT^+) for survival. Brief exposures to higher concentrations of TPTH for 24, 48, and 72 hours followed by 30 day recovery periods caused significant reductions in survival and growth at 13.0, 13.0, and 6.0 μg/L (12.4, 12.4, 5.7 μg/L TPT^+) respectively. The estimated 30-day LC_{50}'s for the 24h, 48h, 72h and 30 day exposures were 16.4, 8.6[2], 3.9 and 1.5 μg/L respectively (15.6, 8.2, 3.7, 1.4 μg/L TPT^+). An acute-chronic ratio of 31 for TPTH on fathead minnow larvae can be calculated based on the 96 hour lethal and 30 day sublethal toxicity concentrations. The study by Jarvinen et al. (1988) is considered a primary study.

In the study by de Vries et al. 1991 on rainbow trout yolk sac larvae (see Section 2.2.1.3 and 2.2.1.1), 25% mortality and reduced resistance of the survivors to bacterial infection were observed after continuous exposure to 0.6 nM TPTC (0.21 μg/L TPT^+) for 110 days. The NOEC for both endpoints was 0.12 nM TPTC (0.042 μg/L TPT+). Melanization of fish tails, decreases in liver glycogen and increases in liver weight relative to body weight occurred at 3 nM TPTC (1.05 μg/L TPT^+). The NOEC for these three endpoints was 0.6 nM TPTC (0.21 μg/L TPT^+). Because nominal test concentrations were reported, the data from the study are considered secondary.

2.2.2 Invertebrates

2.2.2.1 Di-*n*-butyltin

In a study by Holwerda and Herwig (1986), duck mussels (Anodonta anatina) were exposed to DBTC (15 μg/L Sn, 29.4 μg/L DBT^+) for seven months. Mortality was

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[2] This result was reported by the authors as being unreliable.

25% over the entire period. At the end of the exposure period, tin was localized in the kidney, thus indicating excretion as the likely route of depuration. Exposed clams exhibited significant decreases in dry weight (44%) and energy stores (tissue carbohydrates reduced by 77%) when compared to controls. Elevated end products of anaerobic metabolism (lactate, succinate, acetate and propionate) suggested a dependence on an anaerobic form of energy metabolism on exposure to DBT. DBT blockage of the pyruvate dehydrogenase enzyme complex was proposed as responsible for these effects. The study by Holwerda and Herwig (1986), is considered secondary because nominal test concentrations were used.

2.2.2.2 Triethyltin

No chronic studies are available on TET toxicity to freshwater invertebrates.

2.2.2.3 Tributyltin

In 21-day exposures of Daphnia magna to TBT, Brooke et al. (1986) found a significant decrease in reproductive rates (young/adult and young/adult/day) at TBT concentrations of 0.2 µg/L TBT^+ and greater. Survival was affected at exposure concentrations of 0.5 µg/L and greater. The maximum acceptable toxic concentration (MATC) was calculated at 0.14 µg/L. Based on this value and an acute 48h-EC_{50} of 4.3 µg/L, an acute to chronic ratio of 30.71 was calculated. The study by Brooke et al. (1986) was considered to yield primary data.

Ritchie et al. (1974) reported that continuous exposure of adults of the freshwater snail Biomphalaria glabrata to 10 µg/L TBTO (9.7 µg/L TBT^+) completely inhibited oviposition, and killed all snails in 2-5 days. Exposure to 1.0 and 0.1 µg/L (0.97 and 0.097 µg/L TBT^+) reduced oviposition by 90% and 50% respectively. Development of eggs was eliminated at 10 µg/L (9.7 µg/L TBT^+ and reduced by 65% at 1 µg/L (0.97 µg/L TBT^+). In snails exposed to TBTO from hatching, a mortality rate of 60% was observed during 85d exposure at 0.1 µg/L (0.097 µg TBT^+). Oviposition was significantly reduced at 0.001 µg/L (9.7×10^{-4} µg/L TBT^+) while growth was retarded at 1 and 0.1 µg/L (0.97 and 0.097 µg/L TBT^+). Since most of the concentrations used in this study were below analytical detection limits, nominal concentrations were reported. This study is therefore classified as secondary even though the test methodology was reliable.

In a study on the third instar of the mosquito Aedes aegypti, Sherman and Hoang (1981) found reduced success in pupation and retarded development after 9 days exposure to 20 µg/L TBTF (18.8 µg/L TBT^+). None of the larvae matured when exposed to 100 or 200 µg/L TBTF (94 or 188 µg/L TBT^+). This study used nominal concentrations without solution renewal and is considered secondary.

2.2.2.4 Triphenyltin

In the only study available on the chronic toxicity of triphenyltin to freshwater invertebrates, van der Maas et al. (1972) observed a decrease in growth, egg production and successful development of embryos in the pond snail Limnaea stagnalis exposed to 2 µg/L TPTH (1.9 µg/L TPT^+) for five weeks. Higher test concentrations (10 and 50 µg/L TPTH) resulted in 100% mortality. Since little information is offered on experimental conditions (static vs. flow-through or with-renewal, nominal vs. measured test conditions etc.), this study was classified as secondary.

2.2.3 Algae and Aquatic Plants

Since exposures are long relative to life cycles and the effects of the growth medium on toxicity are unknown, all algal studies are considered to be chronic and secondary.

2.2.3.1 Di-*n*-butyltin

Using axenic cultures of a freshwater alga Ankistrodesmus falcatus, Wong et al. (1982) found that DBT inhibited primary productivity, as determined by ^{14}C uptake, with a median inhibition concentration (24h-IC_{50}) of 6800 µg/L Sn (1.3 x 10^4 µg/L DBT^+). Dibutyltin was much less toxic than triethyl, tributyl and triphenyltin.

2.2.3.2 Triethyltin (TET)

In the study by Wong et al. (1982), TET was found to inhibit primary productivity of indigenous Lake Ontario algae with a 24h-IC_{50} of 95 µg/L TET^+. Axenic cultures of A. falcatus and Scenedesmus quadricauda were less sensitive to TET with 24h-IC_{50}'s of 200 and 100 µg/L Sn (347 and 173 µg/L TET^+), respectively. TET also inhibited growth/reproduction of A. falcatus as determined calorimetrically during an 8 day exposure. The 8d-IC_{50} for this species was estimated at 100 µg Sn/L (173 µg/L TET^+).

2.2.3.3 Tributyltin (TBT)

Wong et al. (1982) also exposed algae to tributyltin and determined 24h-IC_{50}'s for primary productivity on Scenedesmus quadricauda, Ankistrodesmus falcatus, Anabaena flos-aquae and a mixed culture of indigenous Lake Ontario algae. Values ranged from 13 to 20 µg/L Sn (32 to 49 µg/L TBT^+) for the axenic algal cultures and 3 µg Sn/L (7 µg/L TBT^+) for the mixed culture. An 8d-IC_{50} for growth/reproduction in A. falcatus was determined at 5 µg/L Sn (12 µg/L TBT^+). Growth was inhibited over a concentration range of 111 to 223 µg/L TBTC (99-199 µg/L TBT^+) for blue-green algae and 111-445 µg/L TBTC (99-396 µg/L TBT^+) for green algae (Blanck et al. 1984, Blanck 1986) during 14 day experiments in which the algae were exposed in 250 µL cultures in microtitration plates.

2.2.3.4 Triphenyltin

Wong et al. (1982) also exposed the algal cultures described above to triphenyltin and observed inhibition in both growth and primary productivity. Twenty-four hour IC_{50}'s for primary productivity ranged from 10 to 40 µg/L Sn (29.5 to 118 µg/L TPT^+) for the axenic algal cultures and 2.0 µg/L Sn (5.9 µg/L TPT^+) for indigenous Lake Ontario algae. An 8d-IC_{50} for growth/ reproduction in A. falcatus was determined at 2 µg/L Sn (5.9 µg/L TPT^+).

2.3 Summary of Toxicity Data

2.3.1 Di-*n*-butyltin

Only a few toxicity studies on DBT were available for use in guideline development. Of the two chronic studies reported, the most sensitive response was in the duck mussel (Anodonta anatina) which experienced 25% mortality, a 44% decrease in dry weight and a 77% decrease in energy stores when exposed to 29.4 µg/L DBT^+ for seven months (Holwerda and Herwig 1986). Rainbow trout yolk sac fry showed a mortality of 28% and a variety of effects including bottom swimming, melanization of the tails, reduced body weight, increased liver mass and decreased resistance of survivors to infection when exposed to 186 µg/L DBT^+ for 110 days (de Vries et al. 1991). In the two available acute studies, a 48h-LC_{50} of 766 µg/L DBT^+ was determined for the golden orfe (Blunden and Chapman 1986) and immobilization of Daphnia magna occurred with a 24h-IC_{50} of 690 µg/L DBT^+ (Vighi and Calamari 1985). Primary production in the freshwater alga Ankistrodesmus falcatus was inhibited (24h-IC_{50}) at 1.3×10^4 µg/L DBT^+ (Wong et al. 1982).

2.3.2 Triethyltin

In chronic studies, the most sensitive response to TET was in the carp in which changes in digestive enzyme activities were observed at 2.6 µg/L TET^+ (Buzinova 1977). Significant changes to glycogen metabolism in the brain of carp were observed at 85 µg/L TET^+ (Stroganov et al. 1974). These changes are expected to result in reduced growth in the fish. Larval development of the sturgeon was arrested at the same concentration (Danil'chenko 1977). In acute studies with carp, severe respiratory and locomotory effects were observed at 853 µg/L TET^+ (Stroganov et al. 1974), and 100% mortality at 426 µg/L TET^+ (Buzinova 1977). A median immobilization concentration (24h-IC_{50}) of 187 µg/L TET^+ was determined for Daphnia magna (Vighi and Calamari 1985). In algal studies, TET inhibited primary productivity of indigenous Lake Ontario algae with a 24h-IC_{50} of 95 µg/L TET^+ (Wong et al. 1982).

2.3.3 Tributyltin

In chronic studies, the most sensitive response to TBT was observed in the snail Biomphalaria. Oviposition in this species, exposed to TBTO from hatching, was significantly reduced at 0.00097 µg/L TBT^+ while growth was retarded at 0.097 µg/L TBT^+ (Ritchie et al. 1974). Reduced growth occurred in fathead minnow embryos and larvae exposed to concentrations of 0.08 µg/L TBT^+ for 33 days (Brooke et al. 1986). An acute/chronic ratio of 32.5 was calculated by the authors for this species. A mortality of 25%, an increase in liver weight and a decrease in body weight, liver glycogen content and resistance of survivors to bacterial challenge were observed in rainbow trout yolk sac larvae exposed to 0.17 µg/L TBT^+ for 110 days (de Vries et al. 1991).

In acute studies, TBT was highly lethal to fish with 96h-LC_{50}'s as low as 3.44 µg/L TBT^+ in rainbow trout (Martin et al. 1989) and 2.6 µg/L TBT^+ in fathead minnows (Brooke et al. 1986). Invertebrates also showed high sensitivity to TBT with 48-96h LC_{50}'s as low as 0.5 µg/L TBT^+ for Hydra and 3.7 µg/L TBT^+ for Gammarus pseudolimnaeus (Brooke et al. 1986). Mixed cultures of indigenous Lake Ontario algae exhibited inhibition of primary productivity with a 24h-IC_{50} of 7 µg/L TBT^+ (Wong et al. 1982).

2.3.4 Triphenyltin

In chronic studies, the most sensitive responses to TPT were observed in the fathead minnow and in rainbow trout. Significant decreases in growth and survival of the fathead minnow occurred at 0.22 and 1.9 µg/L TPT^+ respectively (Jarvinen et al. 1988). A mortality of 25% and a reduced resistance of survivors to bacterial infection were observed in rainbow trout exposed for 110 days to 0.21 µg/L TPT^+ (de Vries et al. 1991). Reduced growth, egg production and success in development of embryos were observed in the common pond snail, Lymnaea stagnalis, at 1.9 µg/L TPT^+ (van der Maas et al. 1972).

In acute studies, a 96h-EC_{50} of 3.5 µg/L TPT^+ was observed for abnormal swimming behaviour in fathead minnows (Jarvinen et al. 1988) and a 96h-LC_{50} of 14.3 µg/L TPT^+ was reported for rainbow trout (Tooby et al. 1975). A 48h-IC_{50} of 10.8 µg/L TPT^+ was reported for immobilization of Ceriodaphnia dubia (Kline et al. 1989) and a 50% mortality of the mollusc Bulinus was observed after a 24h exposure to 10.2 µg/L TPT^+ (Webbe and Sturrock 1964). As in the tributyltins, indigenous Lake Ontario algae were the most sensitive algal species reported with a 24h-IC_{50} for primary productivity of 5.9 µg/L TPT^+ (Wong et al. 1982). An 8-day IC_{50} of 5.9 µg/L TPT^+ was determined for growth/reproduction in Ankistrodesmus.

3 BIOACCUMULATION

The accumulation of alkyltin species in freshwater organisms is dependent on their lipophilicity. Octanol/water partition coefficients, an indication of the lipophilic nature of an organic compound, can be used to estimate bioconcentration levels if measured whole fish bioconcentration factors (BCFs) are not available. The log K_{ow}'s for DBT and TET are very low and for TET the value was even negative, (see Table 3.1). The low log K_{ow}'s suggest that these compounds do not accumulate significantly in aquatic biota. This is supported by the few studies (discussed below) that have been conducted on DBT and TET bioaccumulation in fish and duck mussel tissue. Since no whole fish BCFs are available, these log K_{ow}'s are used in developing the Guidelines for DBT and TET, (see Table 3 in Appendices I and II). In contrast, whole fish BCFs have been measured for TBT and TPT. Therefore, for these two organotins the BCF values are used in Guideline development rather than the log K_{ow}'s, (see Table 3 in Appendices III and IV).

3.1 Di-*n*-butyltin

Tsuda et al. (1988a) reported bioaccumulation factors (BCF's) for various tissues/organs of carp. The ranking of log BCF's in organs of this species was liver (1.0) > gall bladder (0.9) > kidney (0.7). Whole-fish bioconcentration factors were unavailable for DBT. Other bioaccumulation factors for DBT were reported by Holwerda and Herwig (1986) who exposed duck mussels (A. anatina) to 15 µg/L DBT^+ for seven months. BCF's ranged from 8.7 in the mantle and mantle-edge to 14.7 in the foot and 1567 in the kidney.

3.2 Triethyltin

Stroganov et al. (1973) exposed carp to ^{117}Sn labelled TETC at 10 and 100 µg/L (8.5 and 85 µg/L TET^+) for up to 45 days. Lipids and glycogen of the liver, brain and red muscle showed the greatest affinity for TET while proteins showed the least. Except for the gall bladder, all organs had 45 day BCF's of less than 61. A whole fish bioconcentration factor was not reported in this study.

Table 3.1 Octanol/Water Coefficients for Organotins

Parameter	Log K_{ow}	References
Di-n-butyltin	1.6 0.97	Wong et al. (1982) Tsuda et al. (1988a)
Triethyltin	-1.8	Wong et al. (1982)
Tributyltin	2.2-3.8	Tsuda et al. (1988a) Jones and Millson (1982) Laughlin et al. (1986)
Triphenyltin	2.8 2.1	Wong et al. (1982) Tsuda et al. (1987)

3.3 Tributyltin

Martin et al. (1989) reported whole-fish bioconcentration factors of 406 and 570 (based on TBT ion concentration, and total Sn concentration, respectively) for rainbow trout exposed for 64 days to 0.51 µg/L TBT^+. TBT ion and its metabolites, (e.g. di-*n*-butyltin, Bu_2Sn^{2+}; *n*-butyltin, $BuSn^{3+}$ and inorganic tin) were present in all tissues examined. The high percentage of metabolites in the liver, gall bladder and bile suggested that hepatic dealkylation and biliary-faecal excretion occur during TBT depuration.

Tsuda et. al. (1988b) exposed goldfish to 1.7-2.2 µg/L TBTO for 14 days and obtained a whole fish BCF of 1230. This value is used to determine the baseline uncertainty factor in the Guideline development process (See Table III-3).

3.4 Triphenyltin

A whole-fish bioconcentration factor for TPT is available from Tsuda et al. (1988b) who exposed goldfish to 3.0-3.6 µg/L TPTC (2,7-3.3 µg/L TPT^+) for 14 days and obtained a whole fish BCF of 257. Based on the measured lipid content of the fish in this study (2.1% - 2.6%), the BCF normalized to a lipid content of 10% can be calculated as 1090.

Dietary levels of TPTH having no significant toxicological effects in experimental animals (e.g. rats, dogs) are sufficiently well known that an allowable daily intake (ADI) has been defined for humans as 0.5 µg/kg/day TPTH (0.48 µg/kg/day TPT^+) by the FAO/WHO Joint Expert Committee on Pesticide Residues (McCollister and Schober 1975). The whole fish equivalency concentration (FEC) calculated to protect a 70 kg angler consuming 25 g of fish per day can be calculated from the formula

$$\begin{aligned} \text{FEC} &= (\text{ADI} * 70 * 0.5 / 25) * 2.5 \qquad \text{(MOE 1992)} \\ &= 1.67\ \mu\text{g/g TPT}^+. \end{aligned}$$

By dividing the whole fish FEC by the whole-fish BCF normalized for 10% lipid content (1090) and applying a safety factor of 10 (MOE 1992), a water concentration designed to protect anglers against health effects from bioaccumulation is calculated as 0.2 µg/L TPT^+.

4 TASTE AND ODOUR

No studies are available on the effects of di-*n*-butyltin, triethyltin, tributyltin or triphenyltin on taste and odour of water and fish tissue.

5 MUTAGENICITY

No data were found on the mutagenic potential of any of the organo-tins to fresh water organisms. However, genotoxic data were available on other organisms.

5.1 Di-*n*-butyltin

Dupont (1983) and Dow (1983) each exposed strains of Salmonella to dibutyltin dilaurate both in the presence and absence of rat liver S9 fraction to provide metabolic activation. In both studies, dibutyltin dilaurate did not induce a positive response in any bacterial tester strain, either with or without metabolic activation. However, using a slightly modified Ames assay, di-*n*-butyltin trioxide (DBTT) was observed to be clearly mutagenic to Salmonella strain TA 100 and it was weakly mutagenic to strain TA98 (Hamasaki et al., 1993). Hamasaki and et al. (1992) also showed DBTT to be positive with the SOS chromotest (a DNA repair assay) in E. coli, although it produced a negative response on the rec test, a DNA repair assay using the rec gene of Bacillus subtilis.

The effect of DBT on metaphase formation in mammalian cells was studied by Jensen et al (1991). They observed that DBT induced abnormal metaphases, inhibited microtubule formation, and was capable of disassembling existing tubules. This could potentially lead to aneuploid cells (cells with an uneven number of chromosomes).

5.2 Triethyltin

No published studies were found on the genotoxic potential of triethyltin.

5.3 Tributyltin

A number of studies were found on the genotoxic potential of tributyltin. Davis et al. (1987) performed an extensive battery of experiments with TBT. Negative results were obtained in the following experiments; the rec assay with Bacillus subtilis, a reverse mutation assay with Klebsiella pneumoniae, a point mutation assay with

Salmonella typhimurium, a gene mutation assay with Schizosaccharomyces panube, a mitotic gene conversion assay with Saccharomyces cereviseae, a sister chromatid exchange assay with Chinese Hamster Ovary cells (with and without S9), and a recessive lethal mutation assay with Drosophila melanogaster. However, the authors found that TBT, at near cytotoxic concentrations, was mutagenic in the Salmonella fluctuation assay with the point-mutant indicator strain, TA100, but not with the frame-shift mutant indicator strain, TA98. Schweinforth and Gunzel (1987) also found that TBTO failed to induce point mutations in the Ames test, or genetic alterations in S. cerevisae. As well, tests with six TBT "esters" did not induce point mutations in the Ames tests. Recently, Hamasaki et al (1993) observed both tri-*n*-butyltin chloride and bis (tri-*n*-butyltin) oxide (TBTO) to be mutagenic using a modified Salmonella protocol with strain TA100 but not strain TA98, supporting the fluctuation data of Davis et al. (1987). In an earlier publication, Hamasaki et al. (1992) showed that both of these compounds induced positive responses on the rec assay with B. subtilis, yet, contrary to the di-*n*-butyltins, these compounds were negative on the SOS chromotest.

The clastogenic (chromosome or chromatid breaking) potential of tributyltins has been studied in both in vitro and in vivo systems. Early studies looked for the induction of chromosome aberrations in CHO cells in vitro and micronuclei in mouse bone marrow cells in vivo. Neither of these two clastogenic results by themselves were adequately confirmed, however, the two assays in combination provide an indication of a potential clastogenic property for tributyltins. Subsequently, Sasaki et al. (1993) tested tri-*n*-butyltin chloride (TBTC) and TBT fluoride as well as TBTO and found them to be non-clastogenic in vitro while all three compounds were co-clastogenic (i.e., they enhance the chromosome breaking properties of clastogens) with a variety of chromosome breaking compounds. Furthermore TBTO and TBTC were shown to enhance chromatid breaks induced by polluted chlorinated water extracts. Yamada and Sasaki (1993) looked at the in vivo clastogenic properties of TBTO and found it to be non-clastogenic by itself but potentially co-clastogenic.

Like the DBTs, TBTs are also inducers of abnormal metaphases, interfere with spindle tubule assembly, and are capable of disrupting preassembled tubules (Jensen et al. 1991). Therefore they are also potential aneuploidic agents.

Non-maternal toxic doses of up to 6 mg/kg TBT in mice did not produce teratogenic effects, however at high doses (18 mg/kg) in rats teratogenesis was evident, mainly in the form of cleft palates. These authors also reported that there was no evidence to suggest TBT resulted in cancer promotion due to genotoxicity.

The USEPA has not evaluated TBT for evidence of carcinogenic potential according to IRIS (1993).

5.4 Triphenyltin

Dunkel et al. (1985, as cited in GENETOX 1993) reported that a histidine reversion test with S. typhimurium gave negative results. These negative Ames assay results were supported by the study of Hamasaki et al. (1993), who in an earlier study also showed tri-phenyltin chloride to be negative on the SOS chromotest and the rec test with B. subtilis. Triphenyltin acetate orally administered to mice from 7d to 13w of age at the maximum tolerated dose (0.464 mg/kg) did not have a significant effect on tumour incidence (Bock 1981). TPT hydroxide (TPTH) produced teratogenic effects in rats via ingestion, however a NOEL could not be determined from the available data. Sasaki et al. (1993) showed that tri-phenyltin chloride (TPTC), TPT acetate, and TPT hydroxide were all non clastogenic in vitro but were co-clastogenic with a variety of clastogens whose modes of action were different. Like TBTO, TPTC also enhances clastogenic activity of polluted chlorinated water extract. Sasaki et al. (1994) further showed that TPTC can act synergistically with other co-clastogens. In vivo studies by Yamada and Sasaki (1993) found that TPTC, although nonclastogenic by itself, enhanced the activity of clastogens. As with the DBTs and TBTs, TPTs are potential inducers of aneuploidy as they were also shown to induce abnormal metaphases and inhibit spindle tubule formation. However, unlike DBTs and TBTs, TPTs do not appear capable of disrupting pre-existing tubules (Jensen et al. 1991).

5.4 Summary

Since there are no mutagenicity studies on freshwater organisms available for any organo-tin, numeric guidelines to protect against such effects could not be derived. However, data from other organisms suggest that some dibutyltins and tributyltins may be weakly mutagenic. Furthermore, although some tributyltins and triphenyltins may have, at best, weak clastogenic potential, they, along with dibutyltins can interfere with microtubule formation which could induce aneuploid cells. Some tributyltins and triphenyltins are also co-clastogenic both in vitro and in vivo. The doses initiating the genotoxic and co-clastogenic effects are either within the range of those doses found to be toxic to aquatic organisms or they are considerably higher. Additionally, there is some evidence of a teratogenic potential for TBT and TPT at high doses. Since no data were found regarding TET, it is not known whether mutagenicity should be a concern.

6 DERIVATION OF WATER QUALITY GUIDELINES

The uncertainty factors and Provincial Water Quality Guidelines (PWQG) were developed according to the procedures specified by the Ontario Ministry of the Environment (MOE 1992) and can be found in detail in Table 3 of Appendices I-IV. Use of nominal rather than measured toxicant concentrations was the main reason for classifying toxicity data used in developing the uncertainty factors as secondary instead of primary. References used for Guideline development are presented in Appendix V. Separate Guidelines were developed for di-*n*-butyltin, triethyltin, tributyltin and triphenyltin. Because of the high uncertainty factor and the very low critical value, the guideline for tributyltin is below analytical detection limits (Maguire et al. 1985).

6.1 Calculation of Final Uncertainty Factor

6.1.1 Di-*n*-butyltin

Based on a measured log K_{OW} value of 0.97 (Tsuda et al. 1986), a baseline uncertainty factor of 1000 was used for DBT guideline derivation (Table I-1). The toxicity values from three chronic studies were used in calculating the final uncertainty factor. These include the secondary values from de Vries et al. (1991) on mortality, altered behaviour, reduced body weight and other effects on rainbow trout, the secondary toxicity values from Holwerda and Herwig (1986) on reduced weight and energy stores in the duck mussel Anodonta anatina and the toxicity results from Wong et al. (1982) on inhibition of primary production in the freshwater alga Ankistrodesmus falcatus. Due to the secondary classification, the first two studies were both assigned calibration factors of 0.7 while the algal study was assigned a value of 0.9. Acute toxicity values used in calculating the final uncertainty factor include those of Blunden and Chapman (1986) on mortality in the golden orfe Leuciscus idus melanotus and Vighi and Calamari (1985) on immobilization of Daphnia magna. Both studies were assigned calibration factors of 0.9. The final uncertainty factor was 357 (Table I-3).

6.1.2 Triethyltin

A baseline uncertainty factor of 1000 was used for TET guideline derivation (Table II-1). This value is based on a log K_{OW} of -1.8 measured by Wong et al. (1982). Three chronic toxicity values for fish, all secondary (calibration factor = 0.7), and one for algae (calibration factor = 0.9) were used in calculating the final uncertainty factor. The chronic toxicity values for fish include those of Buzinova (1977) for digestive enzyme activity in carp and Danil'chenko (1977) for larval development in the sturgeon and loach. The algal study was that by Wong et al. (1982) on primary productivity of indigenous Lake Ontario algae. Because of the lack of acute studies on fish, three chronic toxicity values were substituted to reduce the uncertainty factor. All were secondary and assigned a calibration factor of 0.9. These toxicity values include those of Stroganov et al. (1974) on glycogen metabolism in the carp and Danil'chenko (1977) on larval development in the ruffe and pike. The toxicity value from the acute study by Vighi and Calamari (1985) on the immobilization of Daphnia magna was also assigned a calibration factor of 0.9. The final uncertainty factor was calculated to be 203 (Table II-3).

6.1.3 Tributyltin

A certain amount of controversy exists surrounding the true log K_{OW} value for tributyltins. Reported values range from 2.2 to 3.8 (Jones and Millson 1982, Wong et al. 1982, Laughlin et al. 1986, Tsuda et al. 1988a) and depend on the salinity. Since the highest reported log K_{OW}, regardless of salinity, is 3.8 (Laughlin et al. 1986), the baseline uncertainty factor for tributyltin compounds should be 1000 (Table III-1). However, because the BCF for goldfish was greater than 1000 (Tsuda et al. 1988b), the baseline uncertainty factor is set at 10,000 (MOE 1992).

Chronic toxicity values on fish contributing to the calculation of the final uncertainty factor include those of Brooke et al. (1986) on fathead minnow development, survival and growth (primary; calibration factor = 0.5), deVries et al. (1991) on weight gain in rainbow trout exposed from larvae through to fingerling (secondary; calibration factor = 0.7) and Wester and Canton (1987) on growth in the guppy (primary; calibration factor = 0.5) . Chronic invertebrate toxicity values include those of Brooke et al. (1986) on reproduction of Daphnia magna (primary; calibration factor = 0.5), and Ritchie et al. (1974) on oviposition in the snail Biomphalaria (secondary; calibration factor= 0.7). The toxicity value obtained by Wong et al. (1982) on growth and reproduction in Ankistrodesmus was assigned a calibration factor of 0.9.

Acute toxicity values on fish contributing to the final uncertainty factor include those of Martin et al. (1989) on mortality of lake trout (primary; calibration factor = 0.8) and Brooke et al. (1986) on mortality of channel catfish and fathead minnow (primary; calibration factors = 0.8). Acute toxicity values for invertebrates include the measurements of Brooke et al. (1986) on mortality in Hydra and the amphipod Gammarus (primary; calibration factors = 0.8). The final uncertainty factor for tributyltin is 181 (Table III-3).

6.1.4 Triphenyltin

The only measured log K_{OW} value available for TBT (2.8) is reported by Wong et al. (1982). Based on this value the baseline uncertainty factor for triphenyltin compounds is 1000 (Table IV-1). Four chronic toxicity values were used in calculation of the final uncertainty factor. These include the values of Jarvinen et al. (1988) on growth and survival of fathead minnow larvae (primary; calibration factor = 0.5), the values of de Vries et al. (1991) on lethality and reduced resistance to infection in rainbow trout larvae (secondary; calibration factor = 0.7), the value of van der Maas et al. (1972) on growth and reproduction in the pond snail (secondary; calibration factor = 0.7) and that of Wong et al. (1982) on growth and reproduction of the alga Ankistrodesmus (secondary; calibration factor = 0.9).

Acute toxicity values for fish contributing to the calculation of the final uncertainty factor include those of Jarvinen et al. (1988) on mortality of fathead minnows (primary; calibration factor = 0.8), Tooby et al. (1975) on mortality of rainbow trout (secondary; calibration factor = 0.9) and Mayer and Ellersieck (1986) on bluegill mortality (secondary; calibration factor = 0.9). Acute toxicity values for invertebrates include those by Webbe and Sturrock (1964) on mortality of the snail Bulinus (secondary; calibration factor = 0.9) and Kline et al. (1989) on mortality of Ceriodaphnia (primary; calibration factor = 0.8). The final uncertainty factor for triphenyltin compounds is 103 (Table IV-3).

6.2 Calculation of Guideline Values

6.2.1 Di-*n*-butyltin

The proposed Provincial Water Quality Guideline for DBT is 8×10^{-2} µg/L DBT^{+} (Fig. I-1). This value is based on a final uncertainty factor of 357 and a critical value of 29.4 µg/L DBT^{+} (Table I-3). This critical value was reported for decreased dry weight and metabolic effects in the duck mussel Anodonta anatina, (Holwerda and Herwig 1986).

6.2.2 Triethyltin

The proposed Provincial Water Quality Guideline for TET is 4×10^{-1} µg/L TET^{+} (Fig. II-1). This value is based on a final uncertainty factor of 203 and a critical value of 85 µg/L TET^{+} (Table II-3). This critical value was reported for arrested development of sturgeon yolk-sac larvae on exposure from the time of egg fertilization (Danil'chenko, 1977). (Data from enzyme studies without reported subsequent whole body effects, as in Buzinova (1977), and Stroganov et al. (1974), cannot be used for critical values).

6.2.3 Tributyltin

The proposed Provincial Water Quality Guideline for TBT is 5×10^{-6} μg/L TBT^+ (Fig. III-1). This value is based on the final uncertainty factor of 181 (Table III-3) and a critical value of 0.00097 μg/L TBT^+ (Fig. III-1). This critical value was reported to be the lowest concentration causing reduced oviposition in the snail Biomphalaria (Ritchie et al. 1974).

6.2.4 Triphenyltin

The proposed Provincial Water Quality Guideline for TPT^+ is 2×10^{-3} μg/L (Fig. IV-1). This value is based on the final uncertainty factor of 103 (Table IV-3) and a critical value of 0.21 μg/L TPT^+ (Fig. IV-1). This critical value was reported as causing mortality and reduced resistance of survivors to infection in rainbow trout yolk sac fry during a 110 day exposure (de Vries et al. 1991). The PWQG based on toxicity is two orders of magnitude less than the FEC/BCF calculated from bioaccumulation (2×10^{-4} mg/L TPT^+; Section 3.4).

7 RESEARCH NEEDS

Considerable information is required to promote these guidelines to Provincial Water Quality Objectives. Since no studies yielding primary data for DBT and TET are available, a complete set of chronic primary studies on fish and invertebrates is required. In order to promote TBT to an Objective, primary chronic studies are required for two freshwater fish (one of which must be a cold water species), and a non-cladoceran invertebrate reproduction and early life stage test. In view of the extreme sensitivity of tropical pulmonate snails to TBT, chronic toxicity tests on native Ontario pulmonate snails are of prime importance. In order to promote triphenyltin to an Objective, two primary, chronic studies on freshwater fish (including at least one cold water fish) and two primary studies on invertebrates (one of which must be an early life stage test) are required.

8 OBJECTIVES OF OTHER AGENCIES

Water quality criteria for tributyltin and triphenyltin in freshwater have been published by a variety of the government agencies including the United Nations Food and Agricultural Organization (UN-FAO), the US. Environmental Protection Agency, Environment Canada and the Department of the Environment of the U.K.(Table 7.1). The UN-FAO recommends a "safe-level" concentration for TBT of 0.27 µg/L TBT^+ (Duncan 1980). The US-EPA has proposed chronic and acute water quality criteria of 0.0264 µg/L TBT^+ and 0.149 µg/L TBT^+, respectively (USEPA 1988). The chronic criterion is defined as that concentration which should not adversely affect freshwater organisms and their uses if not exceeded as a four-day average more than once every three years. The acute criterion is designed to protect freshwater organisms provided that the one-hour average concentration does not exceed this value more than once every three years.

Environment Canada has derived an interim guideline for TBT of 0.008 µg/L TBT^+ for the protection of freshwater aquatic life (CCME 1992). This interim guideline was based on a lowest effect concentration of 0.08 µg/L observed by Brooke et al (1986) for growth of fathead minnows over 33 days. Environment Canada also derived an interim guideline of 0.02 µg/L for triphenyltin based on the reduction in the growth of fathead minnows observed by Jarvinen et al. (1988) at 0.22 µg/L in a 30d exposure. In both cases a safety factor of 10 was applied.

The Department of the Environment of the U.K. has set an Environmental Quality Standard (EQS) of 20 ng/L for both TBT and TPT for protection of freshwater fish (Zabel et al. 1988). The EQS for TBT was based on observations of sublethal histopathological and behavioral changes in rainbow trout at 0.2 µg/L (Seinen 1981). Again, a safety factor of 10 was applied. In the absence of chronic data on the effects of TPT on fish, the same EQS was chosen for TPT as for TBT. Because of the observed sensitivity of molluscs to TBT (eg. Ritchie et al. 1974), an EQS of 0.001 µg/L, the same as that recommended for protection of marine life, was chosen for TBT to protect all freshwater species. An EQS of 0.02 µg/L was chosen for TPT for protection of all freshwater species.

At 0.000005 µg/L (TBT) and 0.002 µg/L (TPT), the guidelines proposed in this study are considerably more conservative than those published by these other agencies. The available freshwater database for organotins is not sufficient for the production of Ontario Provincial Water Quality Objectives which use safety factors similar to those of Environment Canada and the U.K. Department of the Environment (See Ontario's

Water Quality Objective Development Process, March 1992, Appendix VI). Instead, Ontario produces Guidelines as interim standards. These are produced using "uncertainty factors" rather than "safety factors", the magnitude of the factor reflecting the degree of uncertainty caused by the lack of sufficient primary data for Objective development. This development process is also included in the Objective development document referred to above. Guidelines can be promoted to Objectives when sufficient primary data become available, (see Section 6 for required data). Although "secondary" data can be used in Guideline development, this classification should not be interpreted as necessarily indicating poor experimental technique apart from the unavoidable use of nominal test concentrations. Such is the case when serial dilutions of a measured concentration are still toxic at concentrations below analytical detection limits.

Table 8.1 Freshwater Criteria for TBT and TPT

Agency	Organotin	Description	Concentration µg/L	Reference
UN-FAO	TBT	Safe level	0.27	Duncan (1980)
US-EPA	TBT	Chronic Water Quality Criterion	0.0264	USEPA (1988)
US-EPA	TBT	Acute Water Quality Criterion	0.149	USEPA (1988)
Environment Canada	TBT	Guideline (Interim)	0.008	CCME (1992)
Environment Canada	TPT	Guideline (Interim)	0.02	CCME (1992)
DOE-UK	TBT	EQS (For Fish)	0.02	Zabel et al (1988)
DOE-UK	TPT	EQS (For Fish)	0.02	Zabel et al (1988)
DOE-UK	TBT	EQS (For all Aquatic Life)	0.001	Zabel et al (1988)
DOE-UK	TPT	EQS (For all Aquatic Life)	0.02	Zabel et al (1988)
Ontario Ministry of the Environment	TBT	PWQG	0.000005	This Document
Ontario Ministry of the Environment	TPT	PWQG	0.002	This Document

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Appendix I: Guideline Derivation Forms for Di-*n*-butyltin

Table I-1: **Physical-chemical properties for Di-*n*-butyltin dichloride (References in Parentheses given in Appendix V)**

TABLE I-1: Physical-chemical properties		
Compound:	**Chemical formula:**	**CAS No.:**
Di-*n*-butyltin dichloride	$(C_4H_9)_2SnCl_2$	683-18-1

Properties	
Molecular weight (MW):	303.83 g/mol
Melting point:	43 °C (1)
Boiling point:	135 °C (1)
Physical state at room temp.:	solid
Dissociation constant (pKa):	3.80 (2)
Liquid density (D):	1.36 g/cm^3
Molar volume (MW/D):	223 cm^3/mol
Vapour pressure (P^s):	N/A
Water solubility (C^s):	N/A
Henry's law (P^s/C^s):	N/A

Octanol-water partition coefficient (K_{ow})	
Available log K_{ow} values:	1.0 (measured) (4)
Final chosen log K_{ow} value:	1.0

Baseline uncertainty factor for guideline development		
IF valid log $K_{ow} < 4.0$, use 1000		
IF valid log $K_{ow} \geq 4.0$, use 10 000		
Baseline uncertainty factor:	1000	(entered on Table I-3)

TABLE I-2: Aquatic toxicity data table for di-n-butyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
VERTEBRATES											
golden orfe *Leuciscus idus melanotus*		48h-LC_{50}	-	-	-	-	-	766	SU	5	AS
rainbow trout *Oncorhynchus mykiss*		28% mortality, behavioral changes melanization of tail, increased liver mass, reduced body weight reduced resistance of survivors to infection in 110d exposure	7.35-7.45	11±2	90-100% saturation	-	96-105	186	FU	61	CS
INVERTEBRATES											
Daphnia magna	<24h old	24h-IC_{50}	7.5	20	≈ 6	-	200	690	SU	2	AS
duck mussel *Anodonta anatina*		44% decrease in dry weight and 77% decrease in stored carbohydrate in 7 month exposure	7.3-8.0	-	6-9	-	95-115	29.4	RU	8	CS
ALGAE											
Ankistrodesmus falcatus		concentration causing 50% reduction in $^{14}C\text{-}HCO_3$ uptake. 24h exposure in CHU-10 medium	-	-	-	-	-	1.3×10^4	SU	9	CS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

Table I-3: UNCERTAINTY FACTOR WORKSHEET

CHEMICAL:	CAS No.	CONCENTRATION UNITS
DI-n-BUTYLTIN	683-18-1	μg/L DBT$^+$

Test Conditions		Species (life stage)	Toxicity End Point	Effect conc.	Data Codes[1]	Data Type[2]	Calibration Factor	Reference & Table I-2 ref. no.
ACUTE	VERTEBRATE	Golden orfe (juvenile)	48h-LC50	766	SU	2°	0.9	Blunden and Chapman 1986 (5)
	INVERT.	*Daphnia magna* (neonates)	24h-IC50	690	SU	2°	0.9	Vighi and Calamari 1985 (2)
CHRONIC	VERTEBRATE	Rainbow trout (sacfry to fingerlings)	subleth effects & 28% mortality in 110d exp.	186	FU	2°	0.7	de Vries *et al.* 1991 (61)
	INVERT.	Duck mussel (adult)	44% dec. in dry wt. after 7 months	29.4	RU	2°	0.7	Holwerda and Herwig 1986 (8)
	PLANT	*Ankistrodesmus falcatus*	50% ^{14}C-HCO$_3$ uptake red'n in 24h	1.3 x 10^4	SU	2°	0.9	Wong *et al.* 1982 (9)

CALCULATION OF FINAL UNCERTAINTY FACTOR:

Since Log Kow < 4.00, The Baseline Uncertainty Factor = 1000

Baseline Uncertainty Factor X Calibration Factors (maximum number = 11)

1000 x 0.9 x 0.9 x 0.7 x 0.7 x 0.9 x ☐ x ☐ x ☐ x ☐ x ☐ x ☐

= 357 FINAL UNCERTAINTY FACTOR (NOT TO BE LESS THAN A VALUE OF 13)

CRITICAL VALUE + FINAL UNCERTAINTY FACTOR = PWQG OR INTERIM GUIDELINE VALUE

= 29.4 + 357 = 0.08 μg/L

[1] Assign 2 DATA CODES, one from each of the following rows:

S = static R = static/renewal F = flowthrough

U = unmeasured nominal conc. M = measured conc.

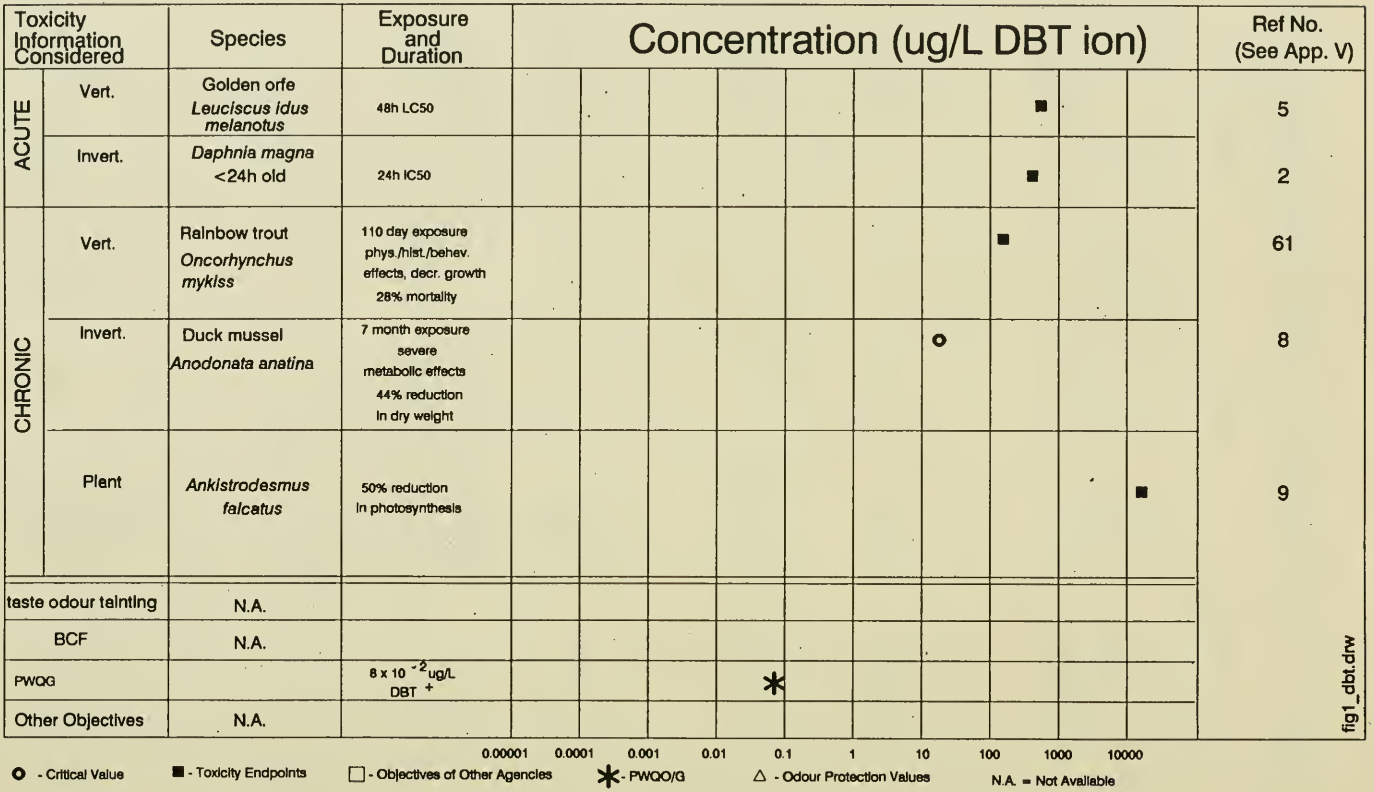
[2] DATA TYPE:

1° = Primary 2° = Secondary 3° = Simulated Data

? = Unknown (Default Data Quality = 2°)

tbl3_dbt.drw

FIG I-1: DERIVATION GRAPH - DI-n-BUTYLTIN



Appendix II: Guideline Derivation Forms for Triethyltin

Table II-1: **Physical-chemical properties for Triethyltin chloride (References in Parentheses given in Appendix V)**

TABLE II-1: Physical-chemical properties		
Compound:	**Chemical formula:**	**CAS No.:**
Triethyltin chloride	$(C_2H_5)_3SnCl$	994-31-0

Properties	
Molecular weight (MW):	241.33 g/mol
Melting point:	15.8 °C (10)
Boiling point:	210 °C (11)
Physical state at room temp.:	liquid
Dissociation constant (pKa):	6.81 (for hydroxide) (12)
Liquid density (D):	N/A
Molar volume (MW/D):	N/A
Vapour pressure (P^s):	N/A
Water solubility (C^s):	N/A
Henry's law (P^s/C^s):	N/A

Octanol-water partition coefficient (K_{ow})	
Available log K_{ow} values:	-1.8 (measured) (9)
Final chosen log K_{ow} value:	-1.8

Baseline uncertainty factor for guideline development		
IF valid log K_{ow} < 4.0, use 1000		
IF valid log K_{ow} ≥ 4.0, use 10 000		
Baseline uncertainty factor:	1000	(entered on Table II-3)

TABLE II-2: Aquatic toxicity data table for triethyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
VERTEBRATES											
carp *Cyprinus carpio*	early juv.	4d-EC$_{100}$ (physiological effects)	-	-		-	-	853	RU	17	AS
"	"	7 day LC$_{100}$	-	-	-	-	-	853	RU	17	AS
"	yearling	3-5d-LC$_{100}$	-	14	-	-	-	426	RU	14	AS
"	early juv.	40 day exposure inhibited brain glycogen formation by 25%	-	-	-	-	-	8.5	RU	17	CS
"	"	40 day exposure inhibited liver glycogen formation by 20%	-	-	-	-	-	85	RU	17	CS
"	yearling	60 day exposure depressed activity of gut/hepatopancreas lipase, amylase and trypsin	-	14-26	-	-	-	2.6	RU	14	CS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE II-2: Aquatic toxicity data table for triethyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ μg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Loach *Misgurnus fossilis*	eggs fertilized in test solution	LOEC for arrested larval development after continual exposure from fertilization	-	-	-	-	-	426	SU	15	CS
Russian sturgeon *Acipenser* sp.	"	LOEC for arrested larval development after continual exposure from fertilization	-	-	-	-	-	85	SU	15	CS
Ruffe *Acerina ceruna*	"	LC_{100} exposure period not reported	-	-	-	-	-	850	SU	16	CS
Pike *Esox lucius*	"	LC_{100} exposure period not reported	-	-	-	-	-	850	SU	16	CS
INVERTEBRATES											
Daphnia magna	< 24h old	24h-IC_{50} in acute immobilization test	7.5	20	-	-	200	187	SU	2	AS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE II-2: Aquatic toxicity data table for triethyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
ALGAE											
Ankistrodesmus falcatus		conc. causing 50% inhibition of primary productivity during 24h exposure in CHU-10 medium	-	-	-	-	-	347	SU	9	CS
"		conc. causing 50% inhibition of growth and reproduction during 8d exposure in CHU-10 medium	-	-	-	-	-	173	SU	9	CS
Scenedesmus quadricauda		conc. causing 50% inhibition of primary productivity during 24h exposure in CHU-10 medium	-	-	-	-	-	173	SU	9	CS
mixed culture of indigenous Lake Ontario algae (species not given)		conc. causing 50% inhibition of primary productivity during 24h exposure in CHU-10 medium	-	-	-	-	-	95	SU	9	CS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

Table II-3: UNCERTAINTY FACTOR WORKSHEET

CHEMICAL:	CAS No.	CONCENTRATION UNITS
TRIETHYLTIN	994-31-0	µg/L TET+

Test Conditions		Species (life stage)	Toxicity End Point	Effect conc.	Data Codes[1]	Data Type[2]	Calibration Factor	Reference & Table II-2 ref. no.
ACUTE	VERTEBRATE	* Carp (early juv.) -	decr. liver glycogen after 40 days	85	RU	2°	0.9	Stroganov *et al.* 1974 (17)
		* Ruffe (eggs/larvae)	LC100	850	SU	2°	0.9	Danil'chenko 1982 (16)
		* Pike (eggs/larvae)	LC100	850	SU	2°	0.9	Danil'chenko 1982 (16)
	INVERT.	*Daphnia magna* (neonates)	24h-IC50	187	SU	2°	0.9	Vighi and Calamari 1985 (2)
CHRONIC	VERTEBRATE	Carp (yearling)	60 day dec. enzyme act.	2.6 **	RU	2°	0.7	Buzinova 1977 (14)
		Russian sturgeon (eggs/larvae)	LOEC (larval devel.)	85	SU	2°	0.7	Danil'chenko 1977 (15)
		Loach (eggs/larvae)	LOEC (larval devel.)	426	SU	2°	0.7	Danil'chenko 1977 (15)
	INVERT.							
	PLANT	Mixed culture of Lake Ontario algae	50% inhib. of 1° prod'n after 24h	95	SU	2°	0.9	Wong *et al.* 1982 (9)

CALCULATION OF FINAL UNCERTAINTY FACTOR:

Since Log Kow < 4.00, The Baseline Uncertainty Factor = 1000

Baseline Uncertainty Factor X Calibration Factors (maximum number = 11)

1000 x 0.9 x 0.9 x 0.9 x 0.9 x 0.7 x 0.7 x 0.7 x 0.9 x ☐ x ☐ x ☐

= 203 FINAL UNCERTAINTY FACTOR (NOT TO BE LESS THAN A VALUE OF 13)

CRITICAL VALUE ÷ FINAL UNCERTAINTY FACTOR = PWQG OR INTERIM GUIDELINE VALUE

= 85 ÷ 203 = 0.4 µg/L

* - Unused chronic data substituted by default as acute data to reduce the uncertainty factor.

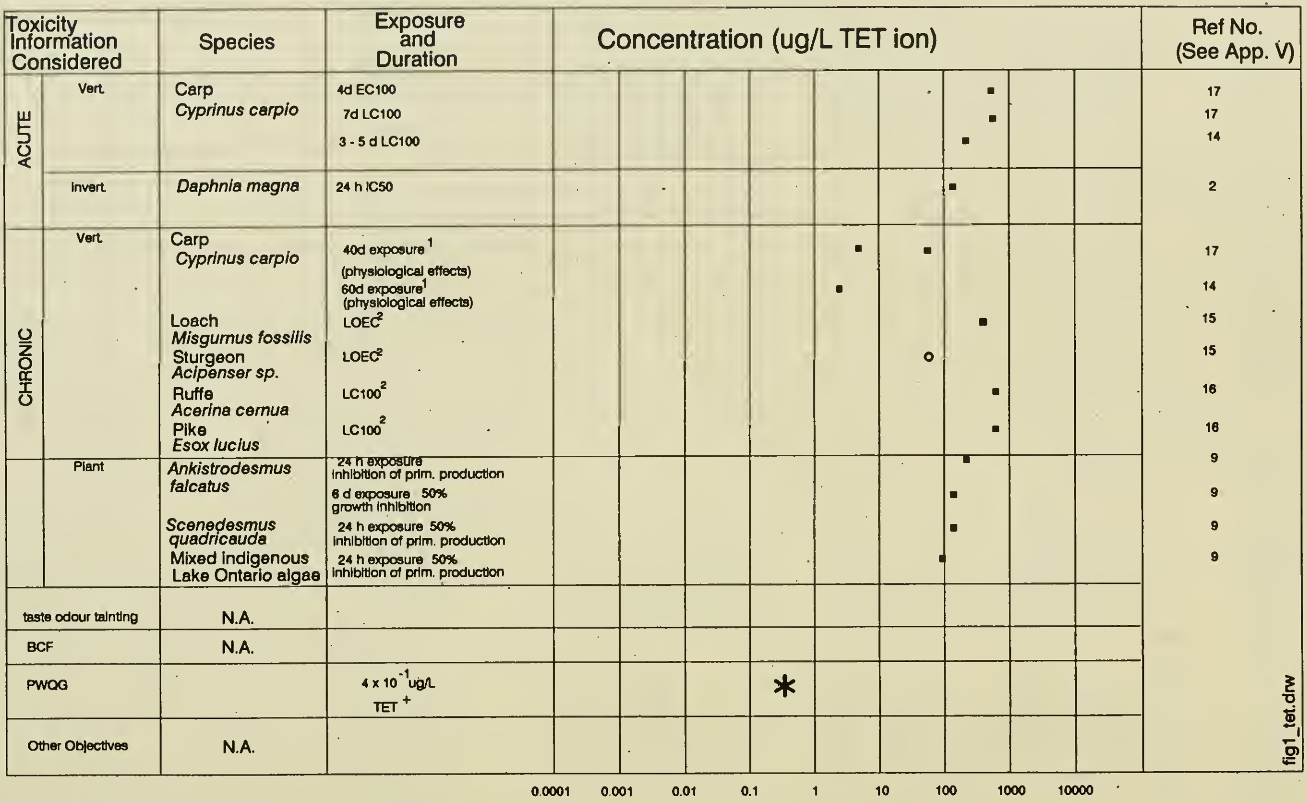
1 - Assign 2 DATA CODES, one from each of the following rows:
S = static R = static/renewal F = flowthrough
U = unmeasured nominal conc. M = measured conc.

2 - DATA TYPE:
1° = Primary 2° = Secondary 3° = Simulated Data
? = Unknown (Default Data Quality = 2°)

tbl3btet.drw

** enzyme effects without whole body effects cannot be used for the critical value

FIG II-1: DERIVATION GRAPH - TRIETHYLTIN



o - Critical Value ■ - Toxicity Endpoints ☐ - Objectives of Other Agencies ✱ - PWQO/G △ - Odour Protection Values N.A. = Not Available

[1] Enzyme effects cannot be used as critical value unless whole body effects are also noted at the same concentration.

[2] Exposed from egg fertilization to larval development (≈ 2 wks).

Appendix III: Guideline Derivation Forms for Tributyltin

Table III-1: Physical-chemical properties for Tributyltin chloride, fluoride, oxide (References in Parentheses given in Appendix V)

TABLE III-1: Physical-chemical properties

Compound:	Chemical formula:	CAS No.:
Tributyltin chloride, fluoride, oxide	$C_{12}H_{27}ClSn/C_{12}H_{27}FSn/$ $C_{24}H_{54}OSn_2$	1461-22-9 (Cl), 1983-10-4 (F), 56-35-9 (oxide)

Properties		
Molecular weight (MW): (chloride)	326.16 g/mol	
Melting point: (fluoride)	248 - 252 °C	(11)
Boiling point: (chloride)	145 - 147 °C	(1)
Physical state at room temp.:	solid	
Dissociation constant (pKa):	N/A	
Liquid density (D): (chloride)	1.20 g/cm^3	(1)
Molar volume (MW/D): (chloride)	272 cm^3/mol	
Vapour pressure (P^s): (TBTO)	0.000085 Pa (= 6.4 x 10^{-7} mm Hg)	(18)
Water solubility (C^s): (TBTO, pH 7.0)	0.007 mol/m^3 (= 4 mg/l)	(18)
Henry's law (P^s/C^s):	212 dyne-cm/g	

Octanol-water partition coefficient (K_{ow})		
Available log K_{ow} values:	2.2 - 3.8	(19,20,21)
Final chosen log K_{ow} value: (TBTO)	3.8	(21)
Bioconcentration factor	1230	(19)

Baseline uncertainty factor for guideline development			
IF valid log K_{ow} < 4.0, use 1000,			If BCF < 1000, use 1000
IF valid log K_{ow} ≥ 4.0, use 10 000			If BCF > 1000, use 10 000
Baseline uncertainty factor:	BCF has precedence	10 000	(entered on Table III-3)

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
VERTEBRATES											
rainbow trout *Oncorhynchus mykiss*	fingerlings	96h LC_{50}	7.8	15.5	8.9	-	135	3.44	FM	22	AP
"	"	24h EC_{50} loss of rheotaxis	7.6	16	7.6	-	350	30.0	FU	23	AS
"	"	damaged gill epithelium cytological abnormalities 21.5h exposure	7.6	16	7.6	-	350	56.4	FU	23	AS
"	"	damaged bile duct epithelium in 5d exposure; corneal damage in 7d exposure	7.6	16	7.6	-	350	11.4	FU	23	AS
"	"	96h LC_{50}	-	-	-	-	-	10.9	FU	5	AS
"	"	96h LC_{50}	-	-	-	-	-	19.4	FU	24	AS
"	"	96h LC_{50}/EC_{50}	7.1	16.4	8.1	41.4	50.6	3.9	FM	25	AP

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity.

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
rainbow trout *Oncorhynchus mykiss*	fingerlings	24h LC_{50}	-	-	-	-	250	27.2	RU	40	AS
"	"	48h LC_{50}	-	-	-	-	250	20.4	RU	40	AS
"	sac fry to fingerling	retarded growth and diminished diminished glycogen storage 110d exposure	-	15	9	-	94-102	0.18	FU	31	CS
"	fry	100% mortality at swim-up after 10-12d exposure	-	15	9	-	94-102	4.5	FU	31	CS
"	sac fry to fingerling	25% mortality; increase in liver mass decrease in body weight, glycogen storage and resistance of survivors to infection in 110d exposure	7.4-7.5	11 ± 2	90-100% satn	-	96-105	0.17	FU	61	CS
lake trout *Salvelinus namaycush*	fingerlings	96h LC_{50}	7.8	15.5	8.9	-	135	12.7	FM	22	AP

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
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F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
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A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
channel catfish *Ictalurus punctatus*		96h LC_{50}	7.1	18.3	7.9	42.5	51.8	5.5	FM	25	AP
fathead minnow *Pimephales promelas*		96h LC_{50}	7.5	24.0	7.4	41.1	51.5	2.6	FM	25	AP
		96h LC_{50}	-	-	-	-	-	49-195	SU	41	AS
"	embryos and larvae	reduced growth with 33 d exposure	7.17	24.2	7.0	42.8	50.4	0.08	FM	25	CP
guppy *Lebistes reticulata*	3-4 weeks	24h LC_{50}	-	-	-	-	-	9.7-19.5	SU	26	AS
"		24h LC_{100}	-	-	-	-	-	73	SU	42	AS
	3-4 weeks	reduced growth/histopathologies in 30d exposure	7.8-8.3	21-25	7.2-8.9	-	-	0.97	RM	6	CP
"		histopathologies in 90 d exposure	7.8-8.3	21-25	7.2-8.9	-	-	0.31	RM	6	CP
"		LC_{50} 7 day exposure (TBTO)	-	-	-	-	-	38	SU	51	CS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
guppy *Lebistes reticulata*		LC_{50} 7 day exposure (TBTC)		-	-	-	-	19	SU	51	CS
Mozambique mouthbrooder *Tilapia mossambica*		24h LC_{50}	7.9	28	-	-	291	27	FU	27	AS
Jewelfish *Tilapia nilotica*		24h LC_{70}	-	-	-	-	-	43.7	SU	43	AS
Tilapia rendalli	fingerlings	24h EC_{50} loss of rheotaxis	7.9	25	7.1	-	350	51.7	FU	23	AS
bluegill *Lepomis macrochirus*		96h LC_{50}	-	-	-	-	-	234	SU	28	AS
medaka *Oryzias latipes*	from freshly fertilized eggs to juvenile	vacuolation of the retina and liver in 3 month exposure	7.8-8.3	21-25	7.2-8.9	-	-	3.1	RU	7	CS

Key 1
TLm = median Lethal Concentration
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Key 2
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F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
golden orfe *Leuciscus idus melanotus*		48h LC_{50}	-	-	-	-	-	48.7	SU	44	AS
common European frog *Rana temporaria*	embryos and larvae	decreased % body water	-	-	-	-	-	29	SU	32	CS
INVERTEBRATES											
Hydra sp.	adult	96hrIC_{50}	7.5	25.9	7.4	42.6	51.0	0.5	FM	25	AP
Oligochaete *Lumbricus variegatus*	0.009 g	96hr IC_{50}	7.1	18.3	7.9	42.5	51.8	5.4	FM	25	AP
Amphipod *Gammarus pseudolimnaeus*	mature	96hr IC_{50}	7.1	18.3	7.9	42.5	51.8	3.7	FM	25	AP
Cladocerans *Daphnia magna*	<24h old	48h IC_{50}	7.5	24.0	7.5	41.1	51.5	4.3	FM	25	AP

Key I
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
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R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Daphnia magna	mature	96h LC_{50}	-	-	-	-	-	5.3	SU	29	AS
"	<24hrs old	24h IC_{50}	7.5	20.0	>6	-	200	11.6	SU	2	AS
"		24h IC_{50}	-	-	-	-	-	68	SU	28	AS
"	<24hrs old	decreased survival and	7.63	20.1	8.5	44.0	53.2	0.5	FM	25	CP
"	"	decreased reproduct. rate in 21 d exp	"	"	"	"	"	0.2	FM	25	CP
Daphnia longispina		24h LC_{100} (TBTO)	-	-	-	-	-	117	SU	42	AS
"		72h LC_{100} (TBTO)	-	-	-	-	-	58	SU	42	AS
"		23h LC_{100} (TBTA)	-	-	-	-	-	249	SU	42	AS
"		72h LC_{100}	-	-	-	-	-	125	SU	42	AS
"		24h LC_{100}	7	-	-	-	-	117	SU	45	AS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
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U = unmeasured toxicant concentration
M = measured toxicant concentratioo

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Ostracod											
Cypridopsis hartwigi		24h LC_{100} (TBTO)	-	-	-	-	-	3900	SU	42	AS
"		48h LC_{100} (TBTO)	-	-	-	-	-	1900	SU	42	AS
"		96h LC_{100} (TBTO)	-	-	-	-	-	116	SU	42	AS
"		24h LC_{100} (TBTA)	-	-	-	-	-	1600	SU	42	AS
"		48h LC_{100} (TBTA)	-	-	-	-	-	830	SU	42	AS
"		96h LC_{100} (TBTA)	-	-	-	-	-	125	SU	42	AS
Cypridopsis hartwigi		24h LC_{100}	7	-	-	-	-	3900	SU	45	AS
Molluscs											
Biomphalaria glabrata (formerly *Australorbis glabrata*	adults	LC_{50} 24h exposure, 48 h recovery	-	25-27	-	-	-	8.3	SU	46	AS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	LC_{100} 24h exposure, 48 h recovery	-	25-27	-	-	-	42	SU	46	AS
"	"	LC_{50} 24h exposure, 24 h recovery	-	-	-	-	-	71	SU	48	AS
"	"	LC_{100} 24h exposure, 24 h recovery	-	-	-	-	-	166	SU	48	AS
"	eggs (1-6 h old)	LC_{50} 24h exposure, 24 h recovery	-	-	-	-	-	26	SU	48	AS
"	eggs (1-6 h old)	LC_{100} 24h exposure, 24 h recovery	-	-	-	-	-	166	SU	48	AS
"	adults	6h LC_{50} (TBTO)	-	-	-	-	-	399	SU	49	AS
"	"	6h LC_{100} (TBTO)	-	-	-	-	-	1900	SU	49	AS
"	"	6h LC_{50} (TBTA)	-	-	-	-	-	241	SU	49	AS
"	"	6h LC_{100} (TBTA)	-	-	-	-	-	831	SU	49	AS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT^+ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	LC_{50} 24h exposure, 72 h recovery	-	23	-	-	-	141	SU	50	AS
	adults	LC_{90} 24h exposure, 72 h recovery	-	23	-	-	-	249	SU	50	AS
"	eggs (5-15h old)	LC_{50} 24h exposure, 72 h recovery	-	23	-	-	-	22	SU	50	AS
"	eggs (5-15h old)	LC_{90} 24h exposure, 72 h recovery	-	23	-	-	-	38	SU	50	AS
"	adults	LC_{100} (TBTO) 24h exposure, 72h recovery	-	-	-	-	-	73	SU	42, 45	AS
"	"	LC_{100} (TBTA) 24h exposure, 72 h recovery	-	-	-	-	-	250	SU	42	AS
"	"	LC_{100} (TBTO) 48h exposure, 72h recovery	-	-	-	-	-	49	SU	42	AS

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect concentration

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT^+ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	LC_{100} (TBTA) 5d exposure, 72h recovery	-	-	-	-	-	125	SU	42	AS
"	"	LC_{100} (TBTO) 5d exposure, 72h recovery	-	-	-	-	-	29	SU	42	AS
"	"	LC_{100} (TBTA) 5d exposure, 72h recovery	-	-	-	-	-	25	SU	42	AS
"	"	24h LC_{50} (TBTA) 24h recovery	-	25	-	-	-	71	SU	47	AS
"	"	24h LC_{90} (TBTA) 24h recovery	-	25	-	-	-	96	SU	47	AS
"	"	6h LC_{50} (TBTA) 24h recovery	-	25	-	-	-	158	SU	47	AS
"	"	6h LC_{90} (TBTA) 24h recovery	-	25	-	-	-	249	SU	47	AS
"	"	24h LC_{50} (TBTO) 144h recovery	-	-	-	-	-	49-97	SU	52	AS
"	"	24h LC_{50} (TBTA) 48h recovery	-	-	-	-	-	83-249	SU	52	AS

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TLm = median Lethal Concentration
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TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	24h LC_{50} (TBTO) 24h recovery	-	25	-	-	-	39	SU	47	AS
"	"	24h LC_{90} (TBTO) 24h recovery	-	25	-	-	-	102	SU	47	AS
"	"	6h LC_{50} (TBTO) 24h recovery	-	25	-	-	-	360	SU	47	AS
"	"	6h LC_{90} (TBTO) 24h recovery	-	25	-	-	-	876	SU	47	AS
"	"	2-5d LC_{100}	-	-	-	-	-	9.7	RU	33	CS
"	<24h to adults	oviposition and growth, respectively reduced during 85d exposure	-	-	-	-	-	0.00097	RU	33	CS
			-	-	-	-	-	0.097	RU	33	CS
Corbicula fluminea	larvae	24h LC_{50}	-	-	-	-	-	2044	SU	28	AS
INSECTA											
Culex sp.	4th instar	96h IC_{50}	7.6	16.8	>7	40.5	51.5	10.2	RM	25	AP
Aedes aegypti	2nd instar	24h LC_{50} (TBTO)	-	26	-	-	-	16.5	SU	30	AS

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TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Aedes aegypti	4th instar	24h LC_{50} (TBTC)	-	26	-	-	-	347	SU	30	AS
"	4th instar	24h LC_{50} (TBTO)	-	26	-	-	-	370	SU	30	AS
"	4th instar	55% mortality during 24h exposure to TBTC. 25% mortality post treatment (5-8 days)	-	26	-	-	-	312	SU	30	AS
"	4th instar	55% mortality during 24h exposure to TBTO. 27% mortality post treatment (5-8 days)	-	26	-	-	-	292	SU	30	AS
"	2nd instar	16% mortality 5-8 days after 24h exposure to TBTO	-	26	-	-	-	16.5	SU	30	AS
"	3rd instar	reduced success in pupation and retarded development in 9d exposure	-	-	-	-	-	18.8	SU	63	CS
"	3rd instar	no maturation	-	-	-	-	-	94	SU	63	CS

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TABLE III-2: Aquatic toxicity data table for tributyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (TBT^+ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
ALGAE											
Ankistrodesmus falcatus		concentration causing 50% reduction in growth/reproduction in 8d exposure			CHU-10 medium			12	SU	9	CS
"		concentration causing 50% inhibition of primary productivity during 24h exposure			CHU-10 medium			49	SU	9	CS
Scenedesmus quadricauda		"			CHU-10 medium			40	SU	9	CS
Anabaena flos-aquae		"			CHU-10 medium			32	SU	9	CS
mixed culture of indigenous Lake Ontario algae		"			CHU-10 medium			7	SU	9	CS
Blue-green algae (variety of species)		inhibition of growth during 14d exposure in 250 µl cultures	-	-	-	-	-	99-199	SU	36,58	CS
Green algae (variety of species)		inhibition of growth during 14d exposure in 250 µl cultures	-	-	-	-	-	99-396	SU	36,58	CS

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TLm = median Lethal Concentration
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Table III-3: UNCERTAINTY FACTOR WORKSHEET

CHEMICAL:	CAS No.	CONCENTRATION UNITS
Tributyltin	1461-22-9 (TBT chloride)	ug/L TBT+

Test Conditions		Species (life stage)	Toxicity End Point	Effect conc.	Data Codes[1]	Data Type[2]	Calibration Factor	Reference & Table III-2 ref. no.
ACUTE	VERTEBRATE	Rainbow trout (fingerlings)	96h-LC50	3.44	FM	1°	0.8	Martin et al. 1989 (22)
		Channel catfish (fingerlings)	96h-LC50	5.5	FM	1°	0.8	Brooke et al. 1986 (25)
		Fathead minnow (juveniles)	96h-LC50	2.6	FM	1°	0.8	Brooke et al. 1986 (25)
	INVERT.	*Hydra* (adult)	96h-IC50	0.5	FM	1°	0.8	Brooke et al. 1986 (25)
		Gammarus pseudolimnaeus (mature)	96h-IC50	3.7	FM	1°	0.8	Brooke et al. 1986 (25)
CHRONIC	VERTEBRATE	Fathead minnow (embryo to larvae)	33d red. in growth	0.08	FM	1°	0.5	Brooke et al. 1986 (25)
		Rainbow trout (sacfry to fingerling)	110d decrease in wt. gain	0.17	FU	2°	0.7	de Vries et al. 1991 (61)
		Guppy (juveniles)	30d red. in growth	0.97	RM	1°	0.5	Wester & Canton 1987 (6)
	INVERT.	*Daphnia magna* (adult)	21d red. in reprod.	0.2	FM	1°	0.5	Brooke et al. 1986 (25)
		Biomphalaria (adult)	oviposition sign. red.	0.00097	RU	2°	0.7	Ritchie et al. 1974 (33)
	PLANT	*Ankistrodesmus falcatus*	8d-EC50 (growth)	12.5	SU	2°	0.9	Wong et al. 1982 (9)

CALCULATION OF FINAL UNCERTAINTY FACTOR:

Since BCF > 1000, The Baseline Uncertainty Factor = 10000

Baseline Uncertainty Factor X Calibration Factors (maximum number = 11)

10000 X .8 X .8 X .8 X .8 X .8 X .5 X .7 X .5 X .5 X .7 X .9

= 181 FINAL UNCERTAINTY FACTOR (NOT TO BE LESS THAN A VALUE OF 13)

CRITICAL VALUE + FINAL UNCERTAINTY FACTOR = PWQG OR INTERIM GUIDELINE VALUE

= 0.00097 + 181 = 0.000005 ug/L

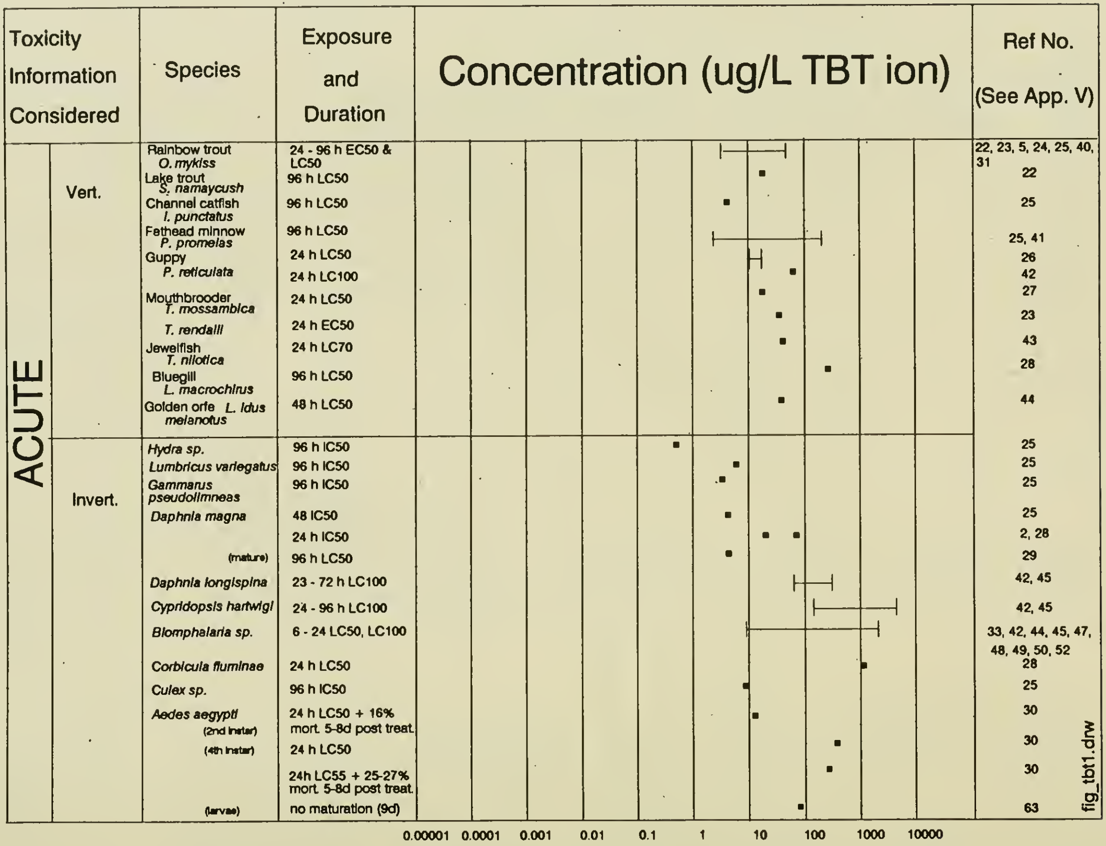
---

[1] Assign 2 DATA CODES, one from each of the following rows:
C = chronic A = acute
S = static R = static/renewal F = flowthrough
U = unmeasured nominal conc. M = measured conc

[2] DATA TYPE:
1° = Primary 2° = Secondary 3° = Simulated Data
? = Unknown (Default Data Quality = 2°)

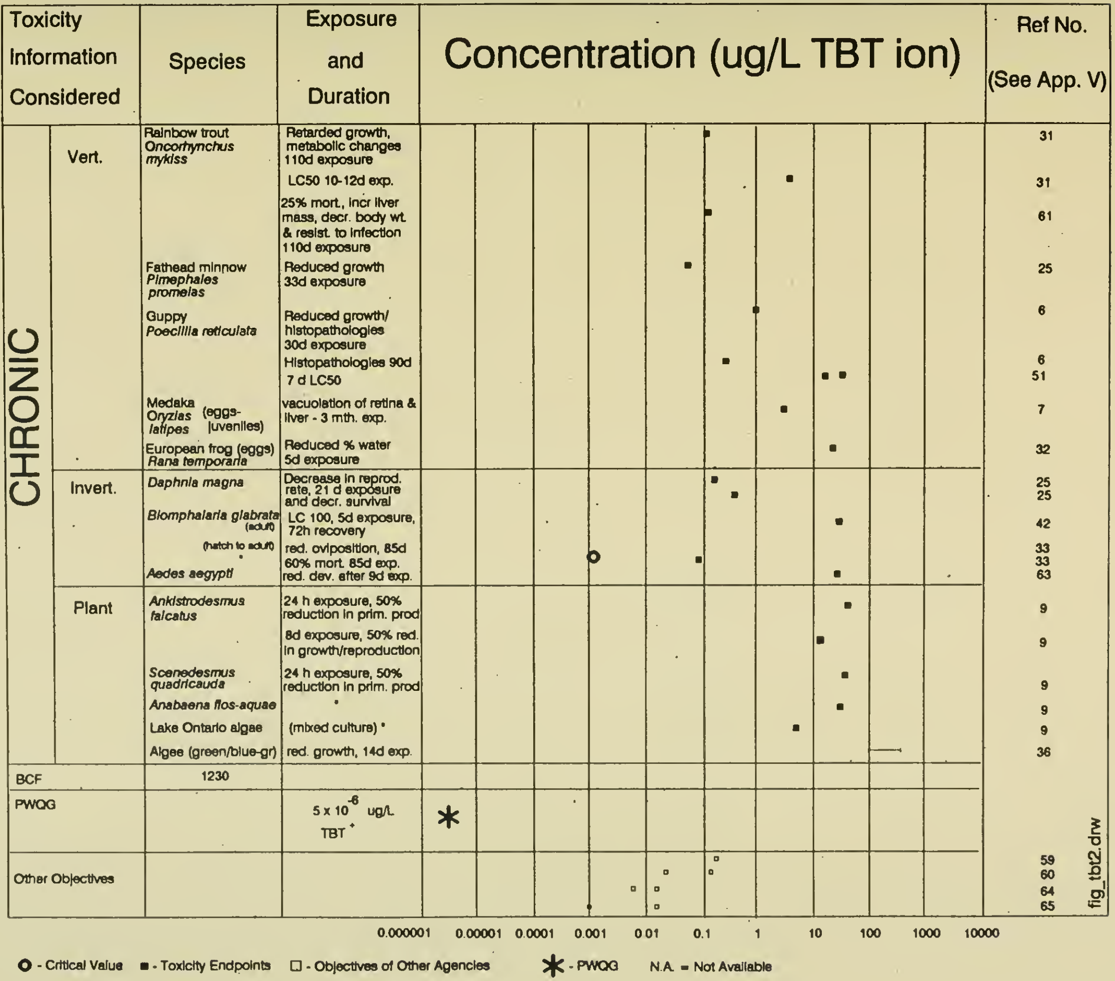
tbl3_tbt.drw

FIG III-1a: DERIVATION GRAPH - TRIBUTYLTIN



Toxicity Information Considered		Species	Exposure and Duration	Ref No. (See App. V)
ACUTE	Vert.	Rainbow trout *O. mykiss*	24 - 96 h EC50 & LC50	22, 23, 5, 24, 25, 40, 31
		Lake trout *S. namaycush*	96 h LC50	22
		Channel catfish *I. punctatus*	96 h LC50	25
		Fathead minnow *P. promelas*	96 h LC50	25, 41
		Guppy *P. reticulata*	24 h LC50	26
			24 h LC100	42
		Mouthbrooder *T. mossambica*	24 h LC50	27
		T. rendalli	24 h EC50	23
		Jewelfish *T. nilotica*	24 h LC70	43
		Bluegill *L. macrochirus*	96 h LC50	28
		Golden orfe *L. idus melanotus*	48 h LC50	44
	Invert.	*Hydra sp.*	96 h IC50	25
		Lumbricus variegatus	96 h IC50	25
		Gammarus pseudolimneas	96 h IC50	25
		Daphnia magna	48 IC50	25
			24 h IC50	2, 28
		(mature)	96 h LC50	29
		Daphnia longispina	23 - 72 h LC100	42, 45
		Cypridopsis hartwigi	24 - 96 h LC100	42, 45
		Biomphalaria sp.	6 - 24 LC50, LC100	33, 42, 44, 45, 47, 48, 49, 50, 52
		Corbicula fluminae	24 h LC50	28
		Culex sp.	96 h IC50	25
		Aedes aegypti (2nd instar)	24 h LC50 + 16% mort. 5-8d post treat.	30
		(4th instar)	24 h LC50	30
			24h LC55 + 25-27% mort. 5-8d post treat.	30
		(larvae)	no maturation (9d)	63

FIG III-1b: DERIVATION GRAPH - TRIBUTYLTIN



Appendix IV: Guideline Derivation Forms for Triphenyltin

Table IV-1: Physical-chemical properties for Triphenyltin hydroxide (References in Parentheses given in Appendix V)

TABLE IV-1: Physical-chemical properties		
Compound:	**Chemical formula:**	**CAS No.:**
Triphenyltin hydroxide	$(C_6H_5)_3SnOH$	76-87-9

Properties		
Molecular weight (MW):	367.02 g/mol	
Melting point:	118 - 120 °C	(1)
Boiling point:	N/A	(2)
Physical state at room temp.:	solid	
Dissociation constant (pKa): (chloride)	5.20	
Liquid density (D):	N/A	
Molar volume (MW/D):	N/A	
Vapour pressure (P^s):	N/A	
Water solubility (C^s):	0.00327 mol/m^3 (= 1.2 mg/L)	(3)
Henry's law (P^s/C^s):	N/A	

Octanol-water partition coefficient (K_{ow})		
Available log K_{ow} values:	2.8 (measured)	(9)
Final chosen log K_{ow} value:	2.8	

Baseline uncertainty factor for guideline development		
IF valid log K_{ow} < 4.0, use 1000		
IF valid log $K_{ow} \geq 4.0$, use 10 000		
Baseline uncertainty factor:	1000	(entered on Table IV-3)

TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)	Effect conc. (DBT^{+} µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
VERTEBRATES											
fathead minnow *Pimephales promelas*	juvenile	96h-LC$_{50}$	7.1	18.0	-	30-35	44	19.1	SM	39	AS
	"	24h-LC$_{50}$	7.1	18.0	-	30-35	44	72	SM	39	AS
"	larvae	96h-LC$_{50}$	6.8-8.0	25	>6	42.0	46.6	6.8	SM	35	AP
"	"	96h-EC$_{50}$	6.8-8.0	25	>6	42.0	46.6	3.5	SM	35	AP
"	"	latent LC$_{50}$ 12h exposure, 84h recovery	6.8-8.0	25	>6	42.0	46.6	59	SM	35	AP
"	"	latent EC$_{50}$ (behaviour) 12h exposure, 84h recovery	6.8-8.0	25	>6	42.0	46.6	48	SM	35	AP
"	"	latent LC$_{50}$ 24h exposure, 72h recovery	6.8-8.0	25	>6	42.0	46.6	19.1	SM	35	AP
"	"	latent EC$_{50}$ (behaviour) 24h exposure, 72h recovery	6.8-8.0	25	>6	42.0	46.6	15.1	SM	35	AP
"	"	latent LC$_{50}$ 48h exposure, 48h recovery	6.8-8.0	25	>6	42.0	46.6	6.2	SM	35	AP

Key 1
TLm = median Lethal Concentration
LC = lethal concentration
IC = immobilization concentration
EC = effects concentration
LOEC = lowest observed effect conc.

Key 2
S = static test
R = renewed static test
F = flowthrough test
U = unmeasured toxicant concentration
M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
fathead minnow *Pimephales promelas*	larvae	latent EC_{50} (behaviour) 48h exposure, 48h recovery	6.8-8.0	25	>6	42.0	46.6	4.8	SM	35	AP
"	"	latent LC_{50} 72h exposure, 24h recovery	6.8-8.0	25	>6	42.0	46.6	5.7	SM	35	AP
"	larvae	latent EC_{50} (behaviour) 72h exposure, 24h recovery	6.8-8.0	25	>6	42.0	46.6	3.3	SM	35	AP
"	larvae to juvenile	decreased growth, 30d exposure	6.8-8.0	25	>6	42.0	46.6	0.22	FM	35	CP
"	"	decreased survival, 30d exposure	"	"	"	"	"	1.9	"	35	CP
"	"	estimated LC50 after 24h exp. & 29d recovery	"	"	"	"	"	15.6	"	35	CP
"	"	estimated LC50 48h exp. & 28d recovery	"	"	"	"	"	≈8.2	"	35	CP
"	"	estimated LC50 after 72h exp. & 27d exposure	"	"	"	"	"	3.7	"	35	CP

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
rainbow trout *Oncorhynchus mykiss*	juvenile	96h-LC_{50}	7.1	13.0	-	30-35	44	26.7	SU	39	AS
	"	24h LC_{50}	7.1	13.0	-	30-35	44	95	SU	39	AS
"	fry	24h LC_{50}	8.1	20	-	-	20	74	RU	37	AS
"	"	48h LC_{50}	8.1	20	-	-	20	29	RU	37	AS
	"	96h-LC_{50}	8.1	15.0	-	-	20	14.3	RU	37	AS
"	sacfry to fingerlings	25% mortality and reduced resistance of survivors to infection in 110d exposure	7.35-7.45	11± 2	90-100% saturation	-	96-105	0.21	FU	61	CS
rainbow trout *Oncorhynchus mykiss*	"	melanization, decrease in liver liver glycogen, increase in liver mass in 110d exposure	7.35-7.45	11± 2	90-100%	-	96-105	1.05	FU	61	CS
carp *Cyprinus carpio*	adult	24h LC_{50}	-	-	-	-	-	1030	SU	57	AS
"	"	48h LC_{50}	-	-	-	-	-	719	SU	57	AS
goldfish *Carassius auratus*	juvenile	96h-LC_{50}	7.1	18.0	-	30-35	44	59	SU	39	AS
		24h LC_{50}	7.1	18	-	30-35	44	115	SU	39	AS

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TLm = median Lethal Concentration
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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT+ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
goldfish *Carassius auratus*	juvenile	24h LC_{100}	6.8	-	-	-	-	342	SU	56	AS
"	"	LC_{100}, 12h exp. & 48h recovery	-	-	-	-	-	72	SU	42	AS
"	"	LC_{100} 24h exp. & 48h recovery	-	-	-	-	-	64	SU	42	AS
"	"	24h LC_{100}	-	-	-	-	-	227	SU	53	AS
bluegill *Lepomis macrochirus*	juvenile	96h-LC_{50}	7.1	24.0	-	30-35	44	21.9	SM	39	AS
"	"	24h-LC_{50}	7.1	24.0	-	30-35	44	72	SM	39	AS
golden orfe *Leuciscus idus melanotus*	juveniles	24h LC_{50}	-	22	-	-	-	154	SU	57	AS
"	"	48h LC_{50}	-	22	-	-	-	94	SU	57	AS
harlequin fish *Rasbora heteromorpha*	juveniles	24h LC_{50}	-	20	-	-	20	359	FU	40	AS
"	"	48h LC_{50}	-	20	-	-	20	188	FU	40	AS
"	1-3 cm TL	24h LC_{50}	8.1	20	-	-	20	59	FU	37	AS
"	1-3 cm TL	48h LC_{50}	8.1	20	-	-	20	40	FU	37	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT^{+} µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
guppy *Lebistes reticulata*	adult	24h LC_{50}	-	25	-	-	-	103	SU	57	AS
"	"	48h LC_{50}	-	25	-	-	-	46	SU	57	AS
"	"	LC_{100} 12h exp. & 48h recovery	-	-	-	-	-	64	SU	42	AS
"	"	LC_{100} 24h exp. & 48h recovery	-	-	-	-	-	26	SU	42	AS
mosquito fish *Gambusia affinis*	adult	24h LC_{100}	6.8	-	-	-	-	342	SU	56	AS
Eel *Anguilla anguilla*		24h LC_{100}	6.8	-	-	-	-	856	SU	56	AS
INVERTEBRATES											
Molluscs *Biomphalaria* sp.	adults	LC_{50} (TPT acetate) 24h exposure, 120h recovery	5.5	-	-	-	-	43	SU	52	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria sp.	adults	LC_{50} (TPT fluoride) 24h exposure, 168h recovery	5.5	-	-	-	-	9.5-522	SU	52	AS
"	"	LC_{100} 24h exposure, 72h recovery	7	-	-	-	-	908	SU	45	AS
"	"	LC_{100} 72h exposure, 72h recovery	7	-	-	-	-	182	SU	45	AS
"	eggs	LC_{100} 24h exposure	7	-	-	-	-	908	SU	45	AS
Biomphalaria sudanica	eggs 2-3 d old	LC_{100} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	214	SU	54	AS
"	4-6 weeks old	LC_{50} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	14.5	SU	54	AS
"	9-11 weeks old	LC_{50} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	32.5	SU	54	AS
Bulinus nasutus	eggs 2-3d old	LC_{100} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	214	SU	54	AS

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M = measured toxicant concentration

Key 3
P = primary
S = secondary
A = acute toxicity
C = chronic toxicity

TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Bulinus nasutus	5.5-8 weeks old	LC_{50} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	10.2	SU	54	AS
"	11-32 week old	LC_{50} 24h exposure, 48h recovery	7.5-7.8	22-24	-	-	-	30	SU	54	AS
Biomphalaria glabrata (formerly *Australorbis glabrata*)	adult	LC_{50} 24h exposure, 48 recovery	-	25-27	-	-	-	43	SU	46	AS
"	adult	LC_{50} (TPTH) 24h exposure, 24h recovery	-	25	-	-	-	157	SU	47	AS
"	"	LC_{50} (TPTA - three suppliers) 24h exposure, 24h recovery	-	25	-	-	-	151-565	SU	47	AS
"	"	LC_{50} 24h exposure, 24h recovery	-	-	-	-	-	564	SU	48	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT^{+} μg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	LC_{100} 24h exposure, 24h recovery	-	-	-	-	-	3420	SU	48	AS
"	"	LC_{50} (TPTA) 6h exposure	-	-	-	-	-	342	SU	49	AS
"	"	LC_{95} (TPTA) 6h exposure	-	-	-	-	-	1200	SU	49	AS
"	"	LC_{50} (TPTC) 6h exposure	-	-	-	-	-	418	SU	49	AS
"	"	LC_{100} (TPTC) 6h exposure	-	-	-	-	-	1800	SU	49	AS
"	"	LC_{50} (TPTA aq) 24h exposure 72h recovery	-	23	-	-	-	428	SU	50	AS
"	"	LC_{90} (TPTA aq) 24h exposure, 72h recovery	-	23	-	-	-	1060	SU	50	AS
"	"	LC_{50} (TPTA - in alcohol) 24h exposure, 72h recovery	-	23	-	-	-	222	SU	50	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT^+ μg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Biomphalaria glabrata	adults	LC_{90} (TPTA - in alcohol) 24h exposure, 72h recovery	-	23	-	-	-	496	SU	50	AS
"	"	LC_{100} 24h exposure, 72h recovery	-	-	-	-	-	908	SU	53	AS
"	eggs 5-7d old	LC_{94} 24h exposure, 120h recovery	-	25-26	-	-	-	856	SU	46	AS
"	"	LC_{71} 24h exposure, 120h recovery	-	25-26	-	-	-	86	SU	46	AS
"	eggs 1-6d old	LC_{50} 24h exposure, 24h recovery	-	-	-	-	-	103	SU	48	AS
"	"	LC_{100} 24h exposure, 24h recovery	-	-	-	-	-	428	SU	48	AS
"	eggs 5-15h old	LC_{50} 24h exposure, 72h recovery	-	23	-	-	-	19	SU	50	AS
"	"	LC_{90} 24h exposure, 2h recovery	-	23	-	-	-	34	SU	50	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT⁺ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
pond snails *Lymnaea stagnalis*		decrease in growth, egg production and embryo development in 5 wk exp.						1.9	SU	62	CS
"		LC100 in 5w exposure	-	-	-	-	-	10.0	SU	62	CS
Cladocerans											
Daphnia magna	<24h old	24h IC_{50}	7.5	20	>70% sat.	-	200	17.2	SU	2	AS
"	"	48h-IC_{50}	7.1-7.8	23.2	-	-	46.5	13.8	SM	38	AP
D. pulex	<24h old	48h-IC_{50}	7.1-7.8	23.2	-	-	46.5	15.7	SM	38	AP
D. longispina		24h LC_{100}	-	-	-	-	-	227	SU	53	AS
		48h LC_{100}	-	-	-	-	-	45	SU	53	AS
		24h LC_{100}	7	-	-	-	-	45	SU	45	AS
Ceriodaphnia dubia	<24 hrs old	48h-IC_{50}	7.1-7.8	23.2	-	-	46.5	10.8	SM	38	AP
Amphipods											
Gammarus fasciatus	mature	48h-LC_{50}	7.1	12.0	-	-	44	63	SU	39	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT$^+$ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
Ostracod *Cypridopsis hartwigi*		24h LC_{100}	7	-	-	-	-	454	SU	45	AS
Insecta *Aedes aegypti*	4th instar larvae	24h-LC_{50}	-	-	-	-	-	381	SU	30	AS
"	2nd instar larvae	24h-LC_{50}	-	-	-	-	-	15.4	SU	30	AS
"	4th instar larvae	54% mortality during 24h exposure and 46% mortality during 5-8 day recovery	-	-	-	-	-	363	SU	30	AS
"	2nd instar	45% mortality during 5-8 day recovery	-	-	-	-	-	15.4	SU	30	AS

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TABLE IV-2: Aquatic toxicity data table for triphenyltin

Species	Lifestage	Response (Key 1)	Test Conditions					Effect conc. (DBT^+ µg/L)	Data code (Key 2)	Ref. # (App V)	Data class (Key 3)
			pH	Temp. (°C)	DO (mg/L)	Alk (mg/L)	Hard (mg/L)				
ALGAE											
Scenedesmus quadricauda		50% inhibition of primary productivity during 24h exposure			CHU-10 Medium			118	SU	9	CS
Ankistrodesmus falcatus		50% inhibition in reproduction and growth during 8d exposure			CHU-10 Medium			5.9	SU	9	CS
"		50% inhibition in primary productivity during 24h exposure			CHU-10 Medium			29.5-118	SU	9	CS
Anabaena flos-aquae		50% inhibition in primary productivity during 24h exposure			CHU-10 Medium			29.5-118	SU	9	CS
Lake Ontario indigenous phytoplankton assemblage		50% inhibition in primary productivity during 24h exposure			CHU-10 Medium			5.9	SU	9	CS

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Table IV-3: UNCERTAINTY FACTOR WORKSHEET

CHEMICAL:	CAS No.	CONCENTRATION UNITS
TRIPHENYLTIN	76-87-9 (TPT hydroxide)	ug/L TPT+

Test Conditions		Species (life stage)	Toxicity End Point	Effect conc.	Data Codes[1]	Data Type[2]	Calibration Factor	Reference & Table IV-2 ref. no.
ACUTE	VERTEBRATE	Fathead minnow (larvae)	96h-LC50	6.8	SM	1°	0.8	Jarvinen et al. 1988 (35)
		Rainbow trout (juvenile)	96h-LC50	14.3	RU	2°	0.9	Tooby et al. 1975 (37)
		Bluegill sunfish (juvenile)	96h-LC50	21.9	SM	2°	0.9	Mayer & Ellersieck 1986 (39)
	INVERT.	*Bulinus nasutus* (juvenile)	24h exp., 48h recovery LC50	10.2	SU	2°	0.9	Webbe & Sturrock 1964 (54)
		Ceriodaphnia dubia (<24h old)	48h-IC50	10.8	SM	1°	0.8	Kline et al. 1989 (38)
CHRONIC	VERTEBRATE	Fathead minnow (larvae)	30d dec. growth	0.22	FM	1°	0.5	Jarvinen et al. 1988 (35)
		Rainbow trout (sacfry to juvenile)	110d - 25% mortality	0.21	FU	2°	0.7	de Vries et al. 1991 (61)
	INVERT.	*Lymnaea stagnalis* (embryo to adult)	5 wk decrease in egg prod., etc	1.9	SU	2°	0.7	van der Maas et al. 1972 (62)
	PLANT	*Ankistrodesmus falcatus*	8d-EC50 (growth)	5.9	SU	2°	0.9	Wong et al. 1982 (9)

CALCULATION OF FINAL UNCERTAINTY FACTOR:

Since Log Kow < 4.00, The Baseline Uncertainty Factor = 1000

Baseline Uncertainty Factor X Calibration Factors (maximum number = 11)

1000 X .8 X .9 X .9 X .8 X .5 X .7 X .7 X .9 X ☐ X ☐ X ☐

= 103 FINAL UNCERTAINTY FACTOR (NOT TO BE LESS THAN A VALUE OF 13)

CRITICAL VALUE + FINAL UNCERTAINTY FACTOR = PWQG OR INTERIM GUIDELINE VALUE

= 0.21 + 103 = 0.002 ug/L

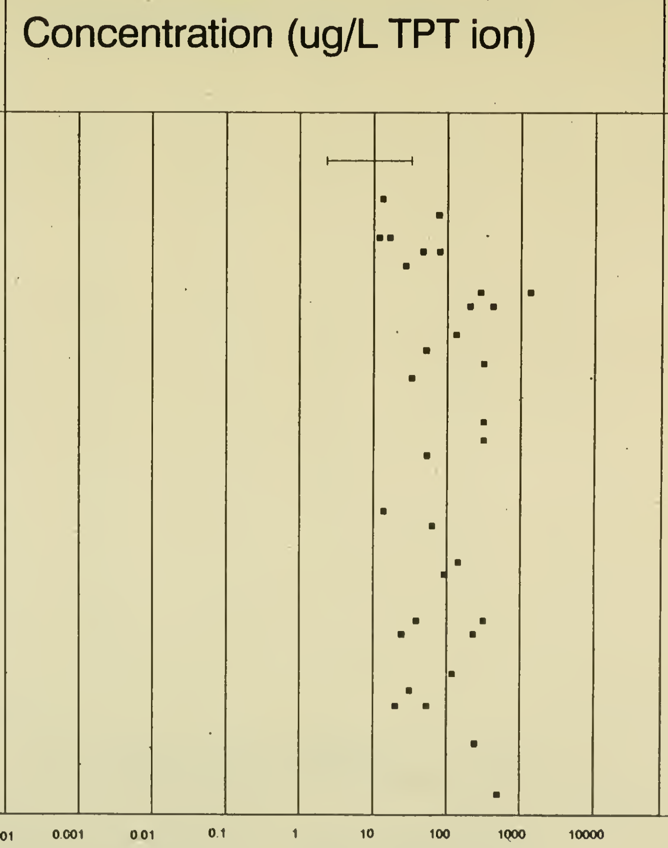
[1] Assign 2 DATA CODES, one from each of the following rows:
C = chronic A = acute
S = static R = static/renewal F = flowthrough
U = unmeasured nominal conc. M = measured conc

[2] DATA TYPE:
1° = Primary 2° = Secondary 3° = Simulated Data
? = Unknown (Default Data Quality = 2°)

tbl3_tpt.drw

FIG IV-1a: DERIVATION GRAPH - TRIPHENYLTIN

Toxicity Information Considered		Species	Exposure and Duration	Ref No. (see App. V)
ACUTE	Vert.	Fathead minnow *Pimephales promelas*	96 h LC50 12 - 96 h exposure	35
			96 h LC 50	39
			24 h LC50	39
		Rainbow trout *Oncorhynchus mykiss*	96 h LC 50	37, 39
			24 h LC 50	37, 39
			48 h LC50	37
		Carp *Cyprinus carpio*	24 h LC50	57
			48 h LC50	57
		Goldfish *Carassius auratus*	24 h LC50	39
			96 h LC50	39
			24 h LC100	56
			LC100 (12 h exposure, 48 h recovery)	42
			24 h LC100	53
			48 h LC100	53
			LC100 (24 h exposure, 48 h recovery)	42
		Bluegill *Lepomis macrochirus*	96 h LC50	39
			24 h LC50	39
		Golden orfe *Leuciscus idus melanotus*	24 h LC50	57
			48 h LC50	57
		Harlequin fish *Rasbora heteromorpha*	24 h LC50	40, 37
			48 h LC50	40, 37
		Guppy *Poacilia reticulata*	24 h LC50	57
			48 h LC50	57
			LC100 (24 h exposure, 48 h recovery)	42
		Mosquito fish *Gambusia affinis*	24 h LC100	56
		Eel *Anguilla anguilla*	24 h LC100	56



fig_tpt1.drw

FIG IV-1b: DERIVATION GRAPH - TRIPHENYLTIN

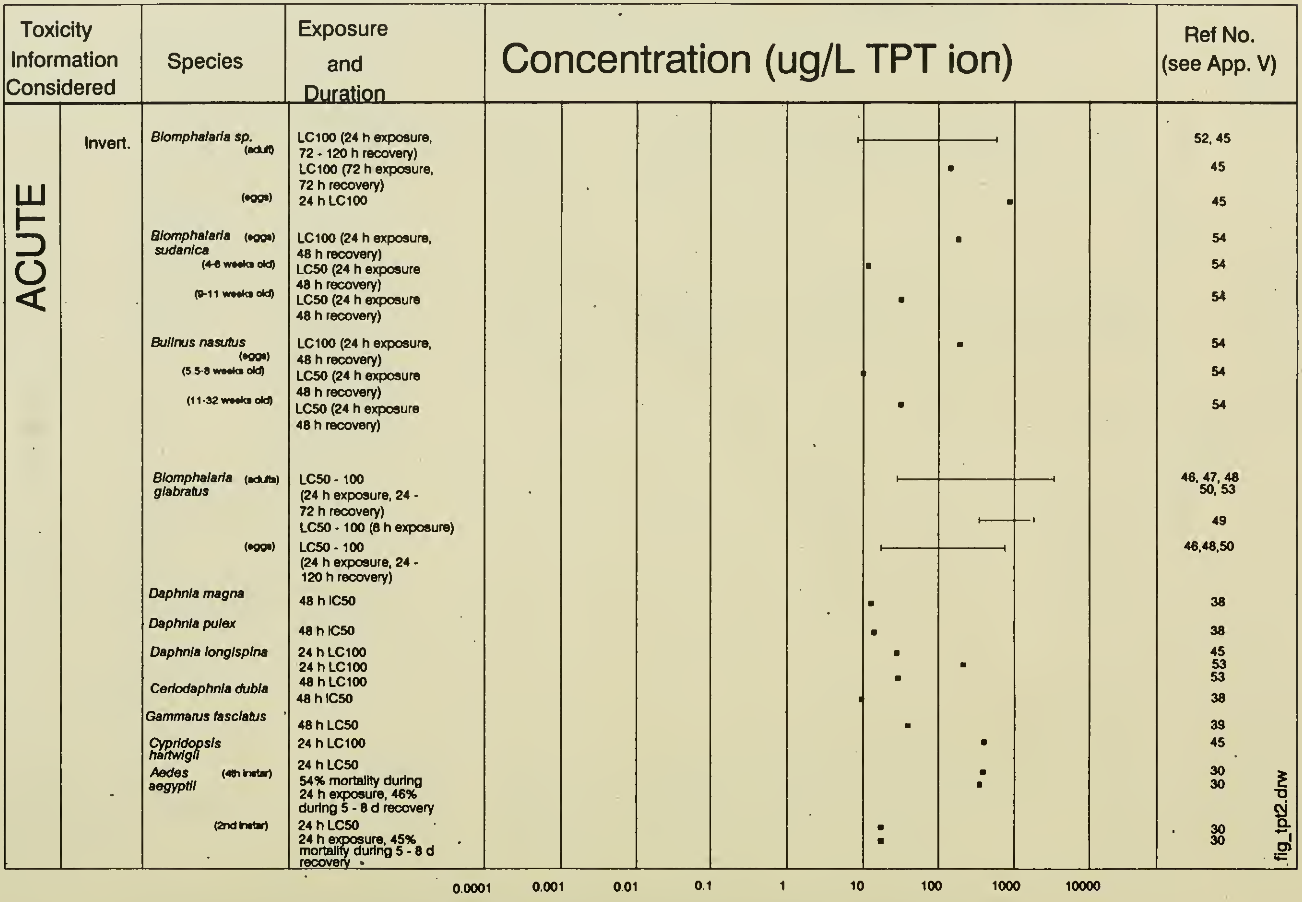
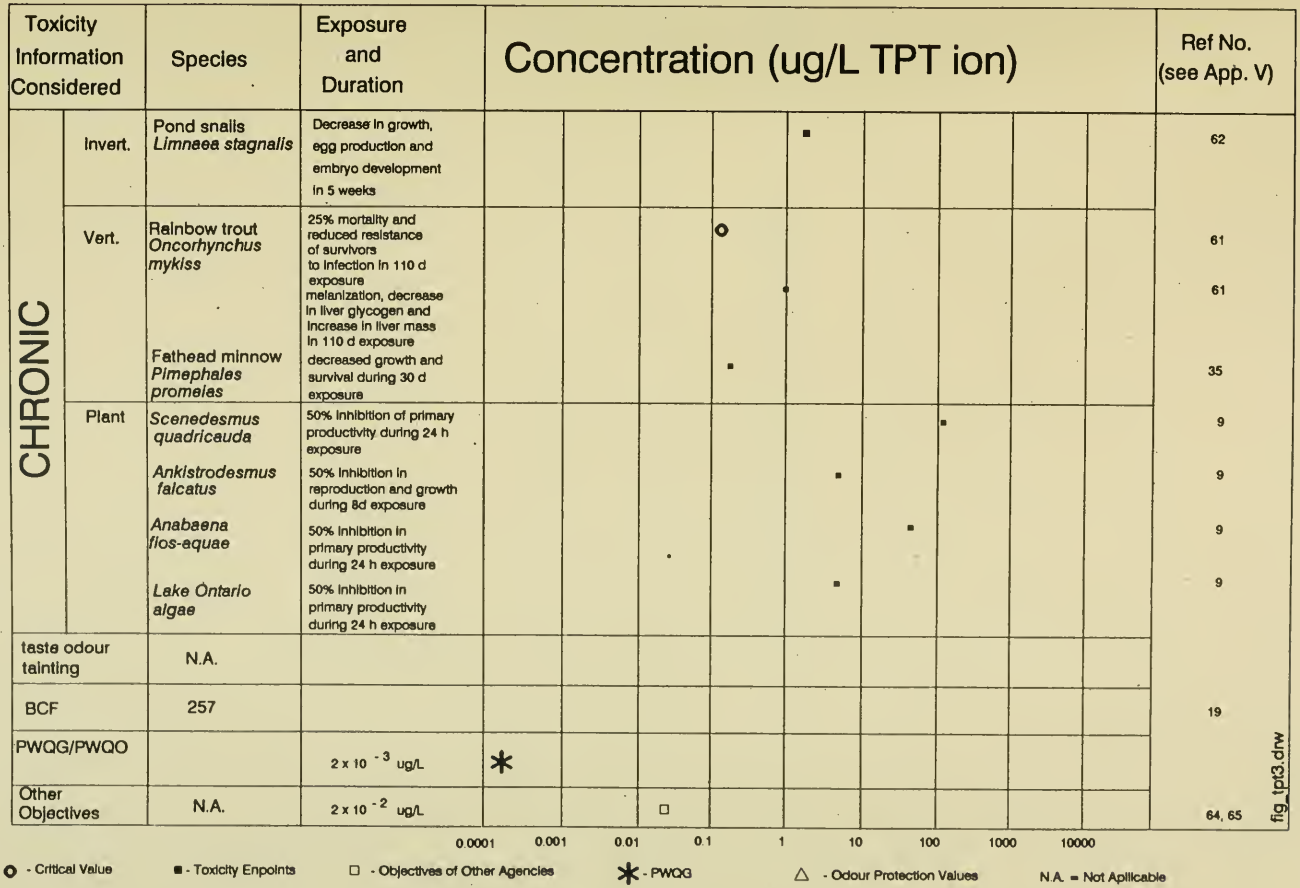


FIG IV-1c: DERIVATION GRAPH - TRIPHENYLTIN



Toxicity Information Considered		Species	Exposure and Duration	Concentration (ug/L TPT ion)	Ref No. (see App. V)
CHRONIC	Invert.	Pond snails *Limnaea stagnalis*	Decrease in growth, egg production and embryo development in 5 weeks	■	62
	Vert.	Rainbow trout *Oncorhynchus mykiss*	25% mortality and reduced resistance of survivors to infection in 110 d exposure	○	61
			melanization, decrease in liver glycogen and increase in liver mass in 110 d exposure	■	61
		Fathead minnow *Pimephales promelas*	decreased growth and survival during 30 d exposure	■	35
	Plant	*Scenedesmus quadricauda*	50% inhibition of primary productivity during 24 h exposure	■	9
		Ankistrodesmus falcatus	50% inhibition in reproduction and growth during 8d exposure	■	9
		Anabaena flos-aquae	50% inhibition in primary productivity during 24 h exposure	■	9
		Lake Ontario algae	50% inhibition in primary productivity during 24 h exposure	■	9
taste odour tainting		N.A.			
BCF		257			19
PWQG/PWQO			2×10^{-3} ug/L	✱	
Other Objectives		N.A.	2×10^{-2} ug/L	□	64, 65

○ - Critical Value ■ - Toxicity Enpoints □ - Objectives of Other Agencies ✱ - PWQG △ - Odour Protection Values N.A. = Not Apllicable

Appendix V: References for Appendices I to IV

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